Lifetime extension of products and Circular Economy. Applications in key sectors for the EU: household appliances and traction batteries.

Doctoral Dissertation

Doctoral Program in Environmental and Civil Engineering (31th Cycle)

# Lifetime extension of products as Circular Economy strategy.

# Applications in key sectors for the EU

Silvia Bobba

* * * * * *

Supervisors

Prof. G.A. Blengini

Prof. Fabrice Mathieux, Co-Supervisor

Doctoral Examination Committee:

Prof. Jo Dewulf, University of Ghent (Belgium)

Prof. Raffaele Vinai, University of Exeter (UK)

Politecnico di Torino

April 30, 2019

I hereby declare that, the contents and organisation of this dissertation constitute my own original work and does not compromise in any way the rights of third parties, including those relating to the security of personal data.

# Abstract


The lifetime of products can be extended through different strategies, e.g. repair, reuse, second-use, which contribute to a more Circular Economy. Accordingly, resource efficiency is maximized and wastage is minimized supporting a more sustainable development. In Europe, reuse of products is attracting the interest of industries in specific sectors and new strategies to extend the lifetime of products are under development.

The extension of lifetime brings potential benefits from environmental, economic and social perspectives, even though such potential benefits have to be verified quantitatively in order to support decision-making and to define what strategies should be incentivized (e.g. design for reuse). A methodological framework to assess the environmental performances of products in a circular economy framework is still work in progress. Currently available indicators for the monitoring of circularity are not able to fully capture the potential environmental benefits of extending lifetime of products. Moreover, potential benefits of reuse strictly depend on the characteristics of the assessed product groups; therefore, the assessment should capture the characteristics of products groups under analysis.

This work contributes to the development of a methodological framework and its implementation to quantify the environmental consequences (benefits or burdens) of extending the lifetime of products in a circular economy context. To capture the complexity of assessing the extension of products lifetime through different strategies, different assessment tools are part of the developed framework and they can be combined in order to provide a wider understanding of the environmental effects of extending products' lifetime. With this aim, the knowledge of the related processes, the knowledge of technical feasibility and sector-specific data are needed. However, due to the novelty of reusing products for certain sectors and also to the existing issues on confidentiality of industrial information/practices, stakeholders should be pro-actively involved in the development of the framework. Therefore, modelling should be coupled with a structured and extensive data collection and a better understanding of the value-chain of products.

In the proposed framework, *Life Cycle Assessment* (LCA) and *Resource Efficiency Assessment of Products* (REAPro) methods are combined to assess the environmental performances of reusing products. According to the developed work, economic and social aspects importantly affect the possibility of extending the lifetime of different products; therefore, the *Environmental and Economic Assessment of Durability of Product* ("Pro-EnDurAncE") was extended to also include economic aspects.
LCA is used to provide the necessary background information of the product/service under analysis, and this is particularly relevant in case of complex systems. The development of modular LCAs and the adoption of parameters make the life-cycle model flexible to update according to available input data and to speed-up the LCAs of different products. The adoption of the same approach allow quick and consistent comparisons between environmental performances of different products/systems. Additionally, due to the fast development of the technology

especially in specific sectors, the modularity of the LCA model allows to enlarge it adding e.g. new materials and/or components.
Finally, *Material Flow Analysis* (MFA) is used to complement the assessment to also capture the potential effects of reusing products in terms of stocks and flows of products and materials. Overall, according to the assessed product sector, one or more methodological components can be combined in order to have a more complete overview of the effects of reusing products in a specific system.

The proposed metrics and framework are applied to two sectors, which are relevant for the EU both economically and environmentally: house appliances and vehicles. The most suitable assessment tools are selected and properly adapted to the sector specificities. For the assessed case-studies, modular LCAs and the adoption of parameters are used to create flexible and customizable models.
For the house appliances sector, the "Pro-EnDurAncE" method is applied to an Energy-related Product, i.e. vacuum cleaner (VC). Lack of data were addressed through the dismantling of a case-study VC and obtained primary data were used for the assessment.
Concerning vehicles, a parametrized and modular LCA model was developed in collaboration with a car manufacturer to estimate the environmental impact of vehicles, to ease the comparison between different vehicles' models and to allow the potential updatability according to the fast development of the mobility. The increasing penetration rate of electric vehicles in Europe is shifting the impacts of vehicles from the use phase to the manufacturing phase, mainly due to the electric powertrain (i.e. electric motor and traction battery); an adapted-LCA and an *ad hoc* energy modelling were used to assess the potential impacts of extending the lifetime of traction batteries in second-use applications.
To track the stocks/flows of batteries in Europe in the next decades and estimate the potential effects of extending the lifetime of batteries, a dynamic MFA model was developed according to the information gathered by the interviewed stakeholders. Parameters allow to assess different scenarios and also different aspects related to batteries, e.g. stocks and flows of storage capacity and embedded materials (in this case Co and Li).
Results of the all the performed analyses pointed out the environmental benefits of extending the lifetime of products entails some environmental benefits under certain conditions and the proposed methodological framework should be adapted to the characteristics of assessed products.

Overall, the developed methodological framework contributes to the field of resource efficiency and offers a framework to assess the environmental effects of extending the lifetime of products. Different stakeholders of the products' value-chain can adopt it to better understand the potential environmental benefits of extending the lifetime of products. The adoption of parameters in all the methodological components allow to update the analysis based on the availability of data. Multiple criteria are used to provide a more complete overview of the impacts (positive and/or negative) of complex systems. The methodological framework can be further extended to include multiple aspects and, if possible, to update models and data through a stricter collaboration with industrial stakeholders.

Keywords:

Lifetime extension; reuse; second-use; Life Cycle Assessment; resource efficiency; Material Flow Analysis; Circular Economy

# Acknowledgment

At the end of this PhD, I would like to thank all the people that supported me during the development of this work.

In particular, I thank Gian Andrea Blengini for pushing me since the beginning to undertake the PhD and for leaving me the freedom to manage it, always providing me his suggestions and recommendations. I thank Fabrice Mathieux for the support in these years at the JRC, for the “scientific meetings” and the encouragement in publishing the results we obtained, always smiling and proving his passion in his work. Also, I thank Fulvio Ardente since my experience as trainee at the JRC: his passion in the work, in finding new ideas and in pushing people in making questions (and finding answers) is an incentive to always go ahead. Thanks to all three also for having supported me in this last months of the PhD, professionally and especially personally.

I thank Isabella for all the suggestions on the PhD procedures, the interesting discussions on life cycle, the creation of lessons and games on LCA for students and for the laughs during all these years.

I would like to thank all the people I met for work reasons. Every meeting I had, I found very enthusiastic people, always open to professional discussions, curious about my work and ready to answer all my questions. I thank JRC colleagues in Petten, especially Andreas Pfrang, Franco Di Persio and Andreas Podias for the SASLAB experience and Marianna Cusenza for the six months spent together in Ispra. I thank prof. Spertino, Alessandro Ciocia and prof. Sambuelli for their support and interest in my research.

Finally, I thank my family for the help, support and enthusiasm in facing all the challenges I met, and especially my parents for always being interested in the development of my work and my life, sharing their ideas and encouraging me. Thanks to my sisters, who helped me in my life choices believing that everything is possible and that I can do it. Thanks to Desirée because she completely overturned our lives and to Alessandro for being with me in living all these experiences.

# Table of contents

# Introduction, policy framework and thesis layout

Products no longer in use, and sometimes when they are still in use, can be replaced by several reasons, for instance when they cannot provide anymore the original service, or when more performant products appear in the market, or they are no more useful, or they are replaced for warranties reasons, etc. Lifetime of products can be extended through their reuse, meant in a larger sense. In fact, products can be repaired through the substitution of specific components or parts when damaged, or they can be remanufactured and used again for the same function; also, they can be repurposed and used in different applications from the previous ones. Hence, lifetime of products depends on several factors.

Historically, money saving was the main driver of reuse. Clearly, reusing products and/or components results in avoiding the usage of new materials to create new products and, as a consequence, money savings but also some benefits for the environment. From the Seventies, awareness on reuse in relation to the environment increased more and more, especially during the Vietnam War, when Americans were claiming about air pollution, waste and water quality. In the following years, the wording “reduce, reuse, recycle” was used as slogan to improve the environmental consciousness of people.

Around the Eighties, waste started to have a positive characterisation, i.e. they were identified as resources. Concepts as “longevity of products”, “life cycle thinking” and “recycling as source of materials” attracted the interest of both business and policy makers (Blomsma and Brennan, 2017). In 1987, the World Commission on Environment and Development (WCED) defined the sustainable development as the “*development that meets the needs of the present without compromising the ability of future generations to meet their own needs*”[1]. Three years later, the term “circular economy” was introduced by the economists Pearce and Turner (Blomsma and Brennan, 2017); only few years later, in 1992, the relevance of considering simultaneously environmental, economic and social aspects in strategic interventions was stressed by the Conference on Environment and Development (UNCED 1992) of the United Nations (UN). As a consequence of this debate on waste and resource management, several approaches arose worldwide in order to codify the links between practices managing waste and resources. In this framework, the Sustainable Development Goals (SDGs) represent the heart of the 2030 Agenda for Sustainable Development, developed thanks to the work of the United Nations and different Countries. SDGs aim at supporting different strategies to “*improve health and education, reduce inequality, and spur economic growth – all while tackling climate change and working to preserve our oceans and forests*”[2].

Efforts towards solutions for a more sustainable development are continuously increasing worldwide, and Europe is a frontrunner in environmental policies (European Commission, 2019;

[1] Brundtland Report, WCED 1987: 43

[2] https://sustainabledevelopment.un.org/?menu=1300

Krämer and Engell, 2017). Also for this reason, this work focuses on the European area and in the following sections, an overview of the development of reuse practices in European policies is provided (section 1.1).

To extend the lifetime of products, several strategies already exist, e.g. repair, reuse, second-use. The adoption of such strategies contributes to improve the resource efficiency and minimize wastage, supporting a more sustainable development. However, in Europe some barriers tackle the fast development of lifetime extension strategies for various reasons (section 1.2). Among these barriers, the lack of a clear and univocal definition of reuse emerged from the performed research (section 1.3).

Moreover, to promote longer lasting products and incentivize measures to drive the extension of the lifetime of products, the sustainability of various options should be proved considering different perspectives and taking into account the technological development according to the specific products sectors. The debate on the potential benefits deriving from the extension of lifetime of products is still open, especially when products are replaced with new and more performant products (Blomsma and Brennan, 2017; Williams and Shaw, 2017). To this aim, potential benefits of reuse should be quantified; however, the quantification of benefits of reuse is still at an early stage even though the attention of both society and manufacturers/primary retail sector is rapidly increasing (Williams and Shaw, 2017). Some assessment tools are already available in the literature; however, they are product-specific and an overall assessment framework is still missing (section 1.4).

## 1.1 Overview of environmental EU policies

In Europe, several actions were launched in the context of sustainability. In the following, an overview of how reuse is addressed in the EU policies is provided.

The consciousness on potential benefits related to a proper end-of-life of products, in particular through their recycling and reuse, increased significantly in the last century. From Nineties, product-oriented environmental policies started to be formulated, led by Netherlands, Denmark and Sweden (Charter et al., 2001). National approaches were then harmonised by the European Integrated Product Policy (IPP), which aims at reducing the "*life cycle environmental impacts of products from the mining of raw materials to production, distribution, use, and waste management*" (EC, 2001). In the Green paper on IPP (EC, 2001), "reuse" is mentioned among the design concepts to pursue resources conservation, reduction of waste, pollution and hazards. Note that the life-cycle perspective is here recognised as a fundamental principle, together with market orientation and stakeholders' involvement. Among the available tools to assess the impacts along the life-cycle of products, the Life Cycle Assessment (LCA) is a standardised tool internationally recognised to assess the environmental performances of products and services. Moreover, even though some methodological aspects are still open, it is recognised as the "*best framework for assessing the potential environmental impacts of products*" available (EC, 2003).

Several policy tools contribute to implement the IPP, e.g. voluntary agreements, standardisation, Eco-design, labelling and product declaration, Greening Public Procurement and legislation, among which waste legislation (e.g. End-of-Life Vehicle Directive and Waste Electrical and Electronic Equipment Directive )[3]. Note that some tools are even prior the IPP Green Paper (Mudgal and Benito, 2008; Shibasaki et al., 2005) Among these, the Energy-using Products (EuP) Directive follows an advanced approach through the development of environmental indicators focusing on the whole life-cycle (Shibasaki et al., 2005).

In order to identify the product groups with the greatest potential for environmental improvement and to promote actions to decrease their environmental impacts, the European Commission (EC) launched the Environmental Impact of PROducts study (EIPRO). Results pointed out that food and drink, private transport (especially passenger cars) and housing[4] are responsible for about 70% of the environmental impact of consumption (Tukker et al., 2006). The contribution of the passenger cars ranges between 15% and 35% according to the assessed impact category, while for housing, the contribution ranges between 20% and 35%. Note that, among products included in the housing, just after the energy use, very important products in terms on environmental impact are energy-using domestic appliances (Tukker et al., 2006).

Focusing on the EU products' policies, in 2008, the EC published the action plan of the Sustainable Consumption and Production and Sustainable Industrial Policy (SCP/SIP) as the natural prosecution of the processes started with the IPP (EC, 2009). The SCP/SIP aims at complementing the already existing policies at both EU and national levels to foster resource efficiency, the adoption of eco-friendly products and the growth of consumers' awareness (Figure 2).

---

[3] http://ec.europa.eu/environment/ipp/toolbox.htm

[4] "Housing" includes buildings, furniture, domestic appliances, and energy for purposes such as room and water heating

Figure 1: Schematic representation of some EU policies related to reuse and lifetime extension since 2008

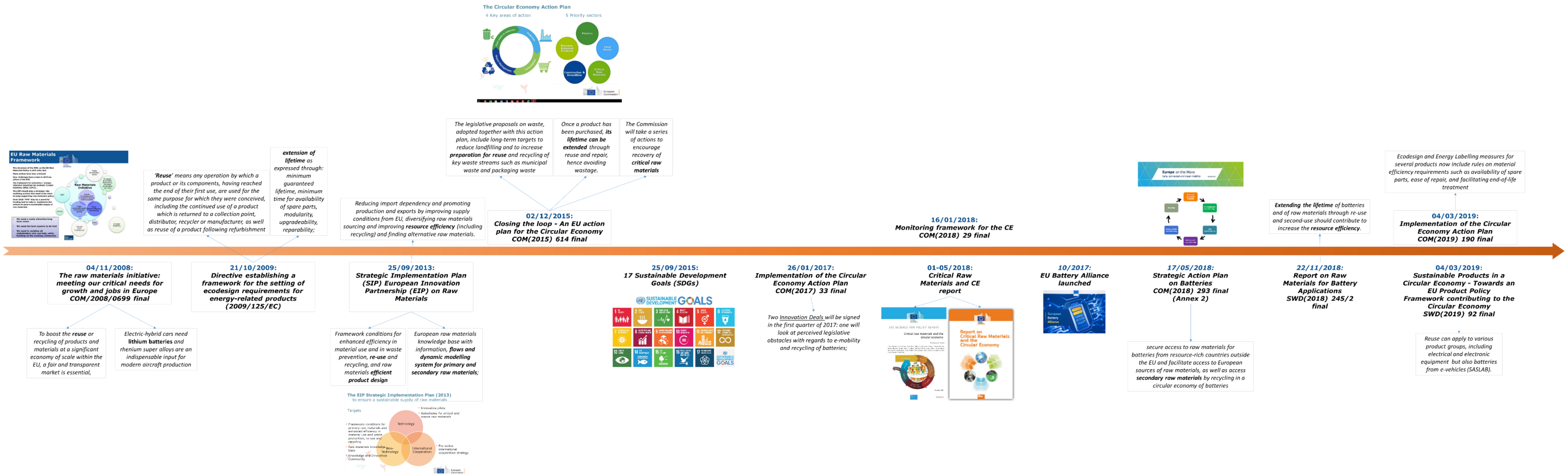
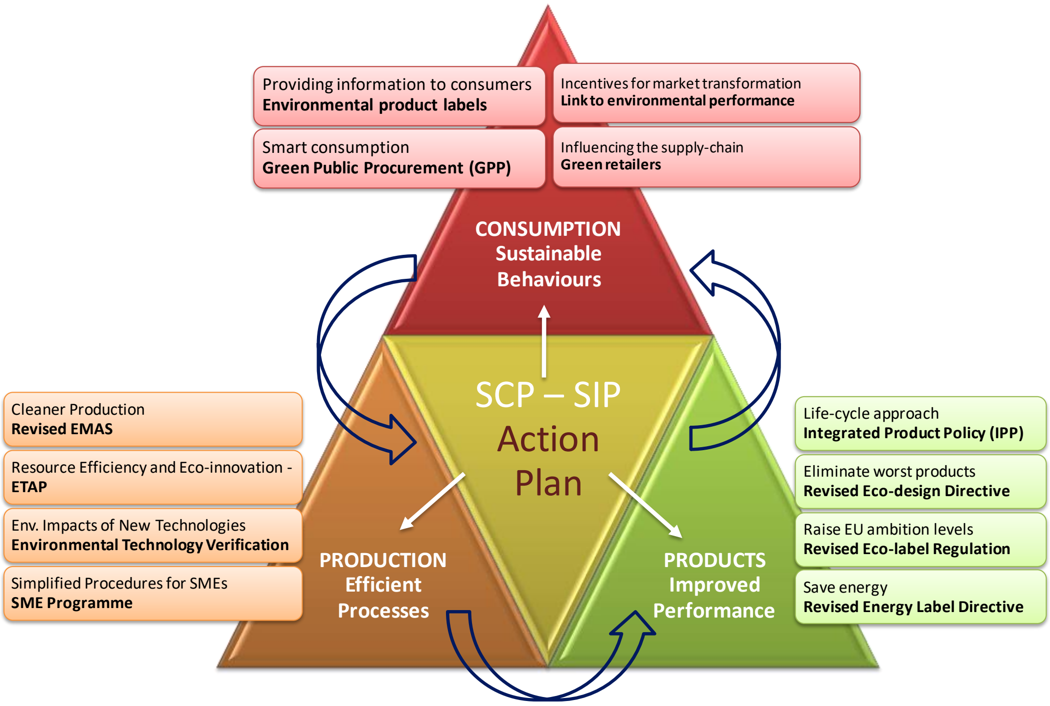


Figure 2: Schematization of policies related to the Sustainable Consumption and Production Action Plan (Mudgal, 2010)




In a life-cycle perspective, waste policies complement the above stated products' policies. As mentioned in the Introduction, the slogan "reduce, reuse, recycle" was already known in Seventies. Then, the wording "reduce, reuse, recycle" was formalised in the Waste Hierarchy in 2008 by the Waste framework Directive (Art. 4 of the Directive 2008/98/EC, now amended by the Directive (EU) 2018/851), where reuse comes before recycling in the waste prevention and management. This means that reuse has higher priority than recycling and recovery. Therefore, Member States are required to take appropriate measures to promote reuse of products and, consequently, to prepare them for reuse activities (also through the adoption of economic instruments, if needed). The promotion of reuse should be implemented using "*educational, economic, logistic or other measures such as support to or establishment of accredited repair and reuse-centres and networks especially in densely populated regions*" (EU, 2008).

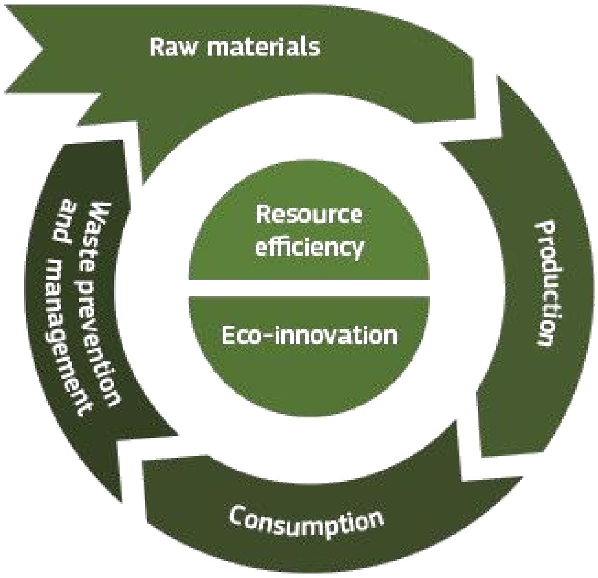
Reuse is mentioned in several other EU policy documents as a key driver to optimise resource and material efficiency, e.g. the Roadmap to a Resource Efficient Europe (EC, 2011a) and the EU's Seventh Environment Action Programme[5]. Such documents also strengthen that reuse should become an attractive option for both public and private actors and that products should be eco-designed for their reuse.

In 2015, the EC adopted a package of measures and legislative proposals to boost sustainable growth and help Europe in making the transition towards a more Circular Economy (CE). A schematic overview of the main concepts of the CE package are represented in Figure 3. It is also

[5] http://ec.europa.eu/environment/action-programme/

noticed that the CE contributes to several SDGs published in 2015 in the 2030 Agenda for Sustainable Development (BOX 1).

Figure 3: Main components of the Circular Economy package
(http://ec.europa.eu/environment/green-growth/index_en.htm)




In such a package, it is stated "*once a product has been purchased, its lifetime can be extended through reuse and repair, hence avoiding wastage. The reuse and repairs sectors are labour-intensive and therefore contribute to the EU's jobs and social agenda. Currently, certain products cannot be repaired because of their design, or because spare parts or repair information are not available*" (EC, 2015). The CE action plan aims at contributing to close the loop of products "*through greater recycling and re-use, and bring benefits for both the environment and the economy*"[6]. Then, the EC promotes reparability, together with upgradability, durability, and recyclability of products.

[6] http://ec.europa.eu/environment/circular-economy/

**BOX 1: Circular Economy and Sustainable Development Goals**

The awareness about the relevance of the environment for both present and future conditions, in line with the Sustainable Development concept, is also reflected by the fact that concerns and challenges for a more sustainable development are discussed at large scale. The Sustainable Development Goals (SDGs) are a result of this concern and discussion at global level (https://sustainabledevelopment.un.org/?menu=1300). In 2015, SDGs were signed by 193 States worldwide, including Europe and the Member States, with the purpose of agreeing on a concrete "to-do list for people and planet".
The EU is contributing to the SDGs with several strategies (https://ec.europa.eu/eurostat/web/sdi), among which the Circular Economy (CE). Focusing on the SDGs and related strategies of the CE to implement it, it is observed that CE contributes to several SDGs.

Graphical representation of the contribution of the CE actions to the SDGs
(based on (European Commission, 2019))

## 1.2 Barriers to reuse

Reuse is not yet fully developed at industrial scale in Europe even though the interest of several industrial sectors is increasing. Pilots and research projects are looking for sustainable solutions to extend the lifetime of products. No doubts that possible solutions should be economic and technical feasible to increase the amount of longer lasting products in the European market. To this aim, some barriers have necessarily to be faced.

According to the developed work, but also consistent with the performed literature review, barriers to reuse can be grouped in 4 groups that are interconnected between them: regulatory and access barriers; technical barriers; market barriers and social barriers. The relevance of these barriers and the existing instruments to face them strictly depends on the product that is considered and on the context in which the products is used. In the following, some important information for the purpose of the PhD work are summarized.

Focusing on different European Countries, it is observed that different approaches to extend the lifetime of products coexist. In many cases, the most common policy instrument is "voluntary agreement"; about one third of the reuse measures relates to information instruments and only 10% of the programmes is implementing regulatory instruments. The remaining 8% refers to economic instruments (EEA, 2018a). Among the regulatory barriers, it emerged that a relevant barrier to implement reuse of products is the absence of a clear wording on "reuse" and related terms at policy/regulatory level (section 1.3).

In addition, technical issues should be faced, especially for complex products. For some products, special skills are required to disassemble and repair, but also to remanufacture and/or repurpose them to then be used in the same application or in second-use applications. In some cases, even though repair operations are feasible, products are not ease to disassemble, and therefore key components of such products cannot be replaced (Cooper, 2010; Kostecki, 1998). Moreover, especially for products belonging to emerging technologies, the technical feasibility of extending the lifetime through various strategies should be proved. Note that the feasibility should also be supported by safety considerations. An example could be the extension of lifetime of Li-ion batteries: technical feasibility is already proved but safety concerns should be faced, yet (e.g. fire risks and toxicity of the electrolyte) (visit to Van Peperzeel B.V. (2017)[7], BOX 8).

The growth of the adoption of reused products also depends on the economic sustainability, and hence on the potential creation of a business case linked to reuse. To support the creation of the business case, a proper management of products after their use, a significant flow of available products for reuse, the necessary infrastructures and skilled workers play a key role (European Union, 2017; Parker et al., 2015). Several aspects participate in the potential increase of the flows of reusable products, e.g. the demand of used products, the availability of used products, economic aspects (e.g. labour price) and physical conditions of products (e.g. good conditions, proper handling) (Cooper and Gutowski, 2015). Some tools related to these aspects are already available. For instance durability and standardization relate to physical factors; whereas warranties and extended producer responsibility[8] relate to the business aspect (Cooper and Gutowski, 2015; EEA, 2018a; European Union, 2017).

For different product categories, the second-hand market occurs in Countries that are different from the Countries of the first use of products. Therefore, reuse of products often used in Europe occurs in third Countries[9] (Baldé et al., 2016). Also, labour costs heavily influence the possibility of the development of a reuse market. The relatively high cost of repairing products compared to the low cost of new products available in the European market negatively impacts on a fast development of a reuse market (Brook Lyndhurst, 2011; Downes et al., 2011; Sabbaghi et al., 2017). These are recognised among the top barriers to reuse (Tecchio et al., 2019). Furthermore, more durable products are generally perceived as more expensive than other products (Downes et al., 2011), which has a relevant impact on consumers' behaviour, and consequently on their choices. Various studies highlighted the relevance of the consumers' behaviour in adopting longer lasting products or in choosing to repair household products instead of replace them. The perception of consumers related to reused and/or longer lasting products is a key role in decision-making (Downes et al., 2011; Sabbaghi et al., 2017); low cost of new products, fashion and low acceptance of users support the replacement of products instead of the adoption of longer lasting products (Cooper, 2005; European Union, 2017; Stahel, 2013).

[7] https://www.peperzeel.nl/

[8] http://www.oecd.org/env/tools-evaluation/extendedproducerresponsibility.htm

[9] For instance e-waste (http://theconversation.com/europes-electronic-waste-has-become-africas-burden-17123). It is estimated that, in 2012, about 0.09 Mt was exported for reuse (Baldé et al., 2016).

In general, consumers' needs (i.e. in terms of functionality, size, appearance, etc.) might change in due course, thus making the extended lifetime and the associated investments obsolete (Ardente and Mathieux, 2014a). A survey carried out to better understand the perception and attitudes on efficient use of resources of people in EU pointed out that about 77% of European citizens claim to prefer repairing their products instead of purchasing new ones, however 39% of the interviewed people think that repairing a broken product is often difficult or economically disadvantageous (TNS, 2014). Therefore, communications programmes to improve the awareness and the availability of highly credible and unambiguous information to consumers are needed to increase acceptability and penetration in the market of more durable products (Bobba et al., 2016a; Cooper, 2005; van Nes and Cramer, 2006).

## 1.3 Definition of reuse

According to the performed work, an unambiguous definition of reuse is still missing. In some cases, this gap represents a barrier for developing reuse in specific sectors (European Union, 2017). Moreover, various terms related to reuse of products are often imprecise or used interchangeably (Ardente et al., 2018; European Union, 2017) (Circusol Workshop (11/03/2019), BOX 2). Similarly, when considering the lifetime of products and its extension through reuse, various "life" terminologies for consumer durables and other machinery are usually used for the same meaning (Murakami et al., 2010). These two concepts are so linked that they are often interchangeably used. The majority of the consulted studies analysed the durability of products focusing on their lifetime extension. However, extending the lifetime not necessarily implies a more durable product, since it is not granted that it will still maintain its performance and functions.

**BOX 2: Circusol Workshop (11/03/2019)**

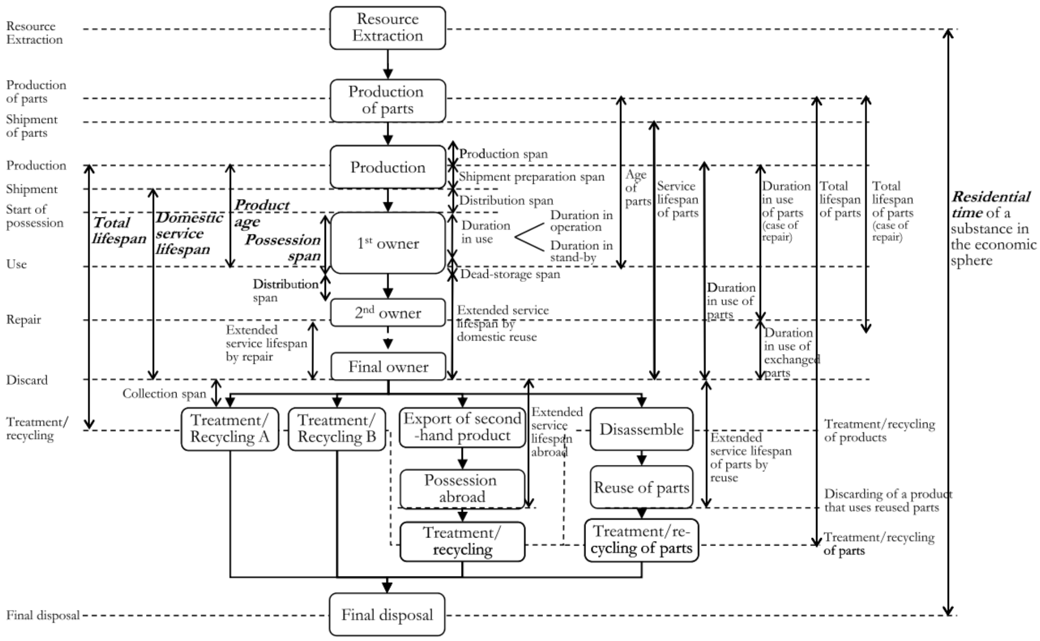
In a recent workshop in Dusseldorf on "Second life of batteries: technical challenges and quality assessment", it clearly emerged that the definition of reuse and related terms differs according to the considered sector. For instance, some participants referred to the definition of reuse as provided by different EU Directives:

- the ELV Directive, "any operation by which components of end-of life vehicles are used for the same purpose for which they were conceived";
- the Waste framework Directive, "any operation by which products or components that are not waste are used again for the same purpose for which they were conceived";
- the WEEE Directive, "any operation by which products or components that are not waste are used again for the same purpose for which they were conceived"

A relevant outcome of the workshop was indeed that all the participants (representing various stakeholders of the batteries value-chain) agreed that a more general and clear definition of reuse and related terms is needed.


Before defining reuse, it is observed that various “life” terminologies are available in the literature. As depicted in Figure 4, lifetime is a fundamental aspect to be considered when studying the stocks and flows of products, as well as their environmental performances in a life-cycle perspective. In fact, the residential time of products could not correspond to their real use and this is particularly relevant in estimating the available flows for recycling but also the impact of products (especially for energy-related products). In the bottom of Figure 4, the reuse options are captured. Note that products but also specific components can change owner and enter in the second-hand market; this makes difficult the estimation of lifetime of products and their traceability along their whole life.

Figure 4: Definitions of various “life” terminologies for consumer durables and other machinery (Murakami et al., 2010)




The common understanding of reuse is that, after its use, a product (or some of its components) can be used again. However, different terms are available in the literature to identify the different types of reuse, e.g. refurbishing, repurposing, second-hand, second-use etc. (Ardente et al., 2018; European Union, 2017; Gharfalkar et al., 2016). Gharfalkar et al. (2016) observed that in most of the assessed studies, it is not possible to clearly understand if the discussed options are part of direct reuse or other types of reuse; in some studies, remanufacturing is not included in reuse, while in other cases, it is considered a specific type of reuse. More in detail, 34% of the studies recognized “repair” or “recondition” or “refurbish” or “remanufacture”’ as a “reuse” option, while the remaining studies do not.

Cooper and Gutowski (2015) schematized the different types of reuse according to some products’ streams (Figure 5). Among the considered waste streams, both appliances and transports, which are recognised two of the most relevant sectors transition towards a resource efficient Europe (European Commission, 2019; Tukker et al., 2006), are included. In both sectors, products can be relocated, i.e. be part of a second-hand market where products perform the same function for which they were perceived, or remanufactured. Remanufacturing includes the repair or replacement of some components (e.g. engine or compressors) in order to keep the products able to perform the same function as during their first life. If this is not possible, products can be adaptive reused for different purposes (adaptive reuse, cascade and reform)[10]; this mainly occurs to the larger components of products, which can be adopted in lower-value applications.

Figure 5: Types of reuse for different products' streams according to (Cooper and Gutowski, 2015)

Business purchases, large items ⟷ Consumer purchases, small items

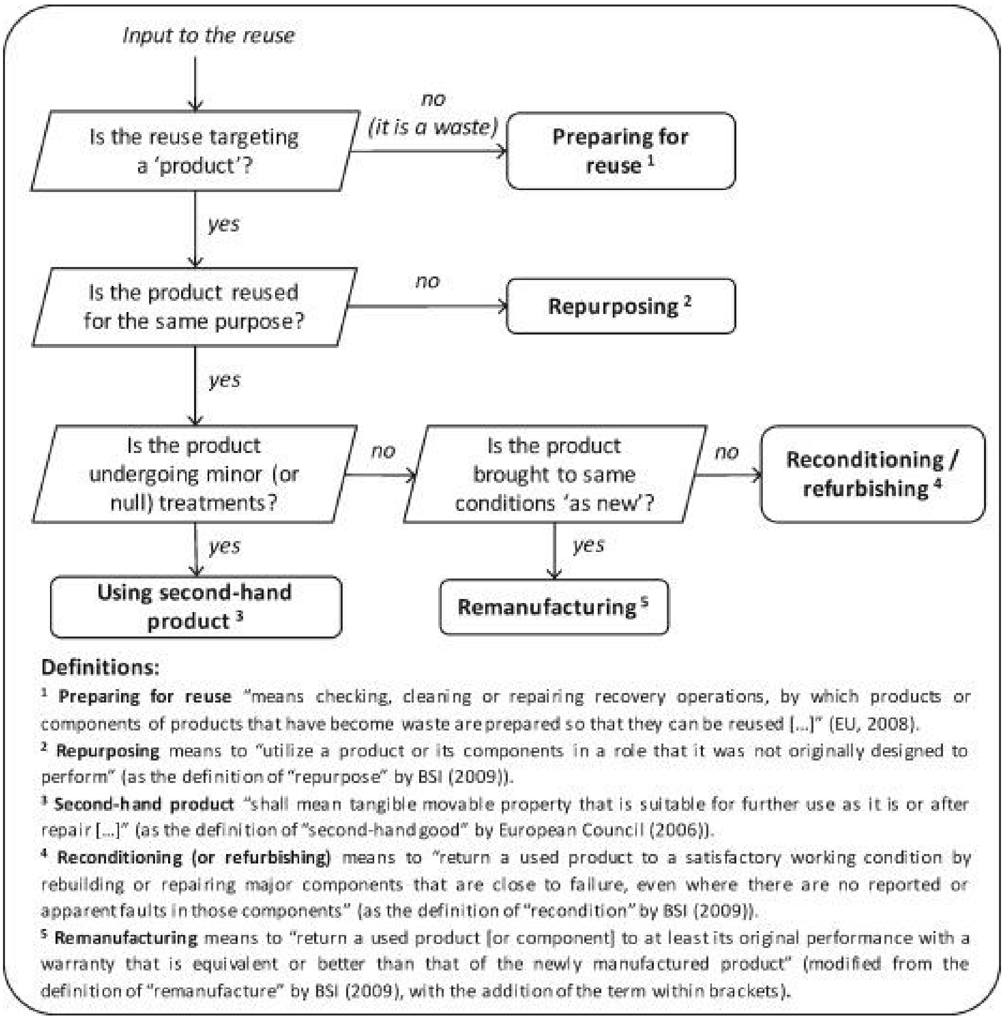
| Reused for… | Reuse activity (applied to products or components) | Construction – buildings and infrastructure | Industrial equipment | Transport | Appliances | Paper and packaging | Textiles |
|---|---|---|---|---|---|---|---|
| …the same purpose: *products & components* | Relocation | Portal frame buildings, steel beams, purlins, or piles[1] | Large equipment such as rolling mills[1] | Used car sales for export[2], Reuse of alloy wheels[3] | Consumer electronics resold on eBay[4] | Shipping containers[1] | Clothing re-sold through charity shops[5] |
| | Remanufacturing: Re-fill | | Liquid gas tanks[6] | | Toner cartridges[7] Domestic propane tanks[25, 26] | Reusable drink containers[8] Refillable containers[6] | |
| | Remanufacturing: Module reuse/ replacement with or without upgrade | BP's North West Hutton oil rig: living quarters were reused onland[1] | Industrial food processing equipment[9] Air-conditioning units[10] | Swedish capped rail system, ReRail[11] Engine remanufacture – recovery (welding) of worn cylinder heads[12, 13] | Photocopier modules[14] Cell phones[15] Remanufacture of refrigerator compressors[16] | | |
| | Remanufacturing: Remediation of component properties | Pre-cast concrete slabs reused in other buildings[17]. | Re-grinding machine tools[10] Industrial electrical equipment, farm equipment, turbines[16] | Aircraft parts, railroad equipment[16] Retreading tires[18] | Lawn mowers[16], white goods[19] | *Toner removal from paper[A]* | |
| | Adaptive reuse | Buildings renovated or extended for a different purpose[1] Reused building foundations[20] | Gas pipe welded to form the supporting truss of the London 2012 Olympic stadium[1] | Discarded tires reused as footwear[21] | *Repurposing of smartphones[F]* | | |
| | Cascade | Reuse of structural steel as shoring in construction[1] and of concrete for sea wave protection (Rip-Rap)[22] | Reuse of steel pipes as building piles[17] | Worn mainline rails are reused on branch-lines[1] | PC chips reused in toys[23] | | |
| …a different purpose: *individual components* | Reform | Ship plate re-rolled into rebar[24] | *Large diameter steel pipe re-rolled into sheet[B], solid bonding of metal machining chips by extrusion[C]* | *Re-forming of car panels[D, E]* | *Re-forming of appliance panels[D, E]* | | |
| | Typical materials: | Steel, aluminum, cement | Metals | Steel, aluminum, plastic | Steel, aluminum, plastic | Paper, steel, alum., plastic | Fabrics |

Current reuse: 1. Allwood et al. (2010a) 2. EP (2010) 3. Cooper and Allwood (2012) 4. Clausen et al. (2010) 5. Well Dressed (2006) 6. Kelle and Silver (1989) 7. Ostlin and Ekholm (2007) 8. Mata and Costa (2001) 9. Butler (2009) 10. Parker and Butler (2007) 11. Rerail (2010) 12. Adler et al. (2007) 13. Venta et al. (1978) 14. Kerr and Ryan (2001) 15. Skerlos et al. (2003) 16. Bollinger et al. (1981) 17. Addis (2006) 18. Ferrer (1997b) 19. Sundin and Bras (2005) 20. Cooper et al. (2014) 21. BBC (2014) 22. FHWA (1989) 23. Geyer et al. (2007) 24. Tilwankar et al. (2008) 25. Ferrellgas (2015) 26. Oliver-Solà et al. (2009) *Prospective reuse: A. Leal et al. (2012) B. Cooper and Allwood (2012) C. Tekkaya et al. (2009) D. Takano et al. (2008) E. Tekkaya et al. (nd) F. Zink et al. (2014)*

As an attempt to clarify definitions related to lifetime extension, Ardente et al. (2018) summarized the different types of reuse considering the "*level of treatment undertaken and the quality of the output*" (Figure 6). The definitions are mainly based on British Standards Institution (BSI), the Directive 2008/98/EC and the Council Directive 2006/112/EC. In the following chapters and sections, the definitions provided by Ardente et al. (2018) are adopted.

[10] In Cooper and Gutowski (2015), cascade refers to the adoption of a product in a different use (e.g., steel pipes can be reused as building piles), while reform refers to components for which the geometry is changed (e.g., the rerolling of ship plate)

Figure 6: Classification and definitions of different types of reuse (Ardente et al., 2018)



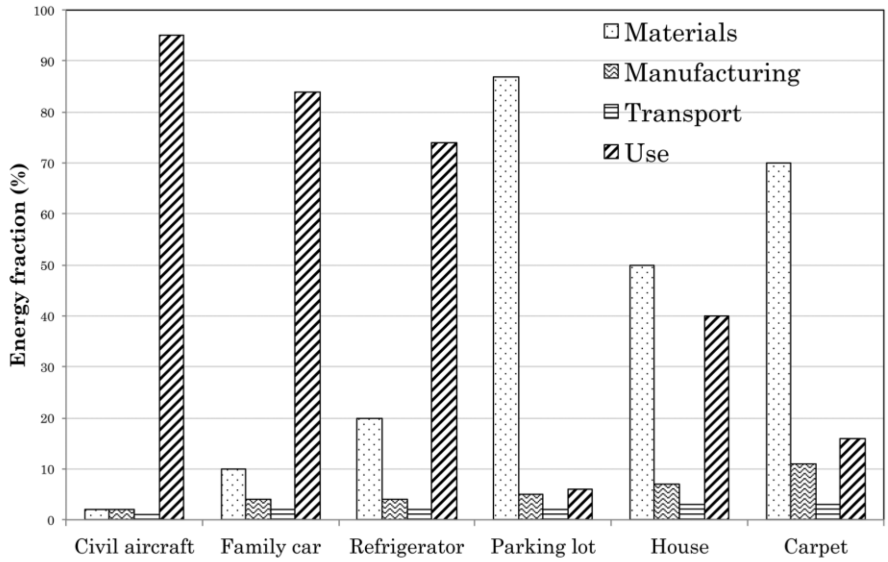

## 1.4 Quantification of benefits of lifetime extension

As above-mentioned, the extension of lifetime of products can contribute to reduce waste generation and save resources. Potential benefits should be quantitatively verified in a life-cycle perspective in order to understand if it is more convenient to repair/remanufacture/repurpose/… products or replace them with new ones.

At the EU level and in the framework of the CE, a monitoring framework was developed to "*to measure progress and to assess the effectiveness of action towards the circular economy in the EU and Member States*" (EC, 2015). Then, a set of indicators was developed to cover four different areas: production and consumption, waste management, secondary raw materials, competitiveness and innovation. Focusing on indicators monitoring End-of-Life (EoL), it is observed that available indicators "*does not capture options higher [than recycling] in the waste hierarchy – repair / reuse / remanufacturing*" (EC, 2018a). Meanwhile, in the document it is highlighted the relevance of monitoring such options especially for products in which Critical Raw Materials (CRMs) are embedded. The recovery of such materials is particularly relevant for the EU since they are materials with a high supply risk and high economic importance (EC, 2017a) (BOX 3). According to Mayer et al. (2018), the effects of extending the lifetime of products on such materials can be captured, for instance, through the estimation of increase/decrease of in-stocks of products.

The debate of the effective benefits related to extending the lifetime of products is still open. According to some studies in the literature, not for all products reuse can have more benefits than recycling since extending the lifetime does not necessarily represent the optimal strategy for some products (Ardente and Mathieux, 2014a; Bobba et al., 2016a; Cooper and Gutowski, 2015; Shibasaki et al., 2005; Sneck, 1981). According to the characteristics of the assessed products, different life-cycle steps can be more/less environmentally relevant. For instance, manufacturing of more durable products could imply the use of higher amount of materials, materials with higher quality or more complex processes, with consequent higher impacts (environmental and economic) (AEA, 2009; Cooper, 1996; Okumura et al., 2001; Planet Ark, 2007). Other authors stressed the relevance of additional features that can imply the product's replacement, as the wear-out of products and the technological evolution of products in the market (Dewulf and Duflou, 2004; Kostecki, 1998; Rose and Stevels, 1999). Focusing on different types of products reported in Figure 7, the use phase is highly significant for vehicles and household appliances, while materials use for manufacturing carpets and houses have the most important contribution to their life-cycle impact (Cooper and Gutowski, 2015). In general, for "unpowered" products the energy efficiency is less relevant than embodied impacts, while is exactly the opposite for energy-related products[11].

Figure 7: Life-cycle contributions for some products (Cooper and Gutowski, 2015)[12]




[11] 'Energy-related product' means *any good that has an impact on energy consumption during use which is placed on the market and/or put into service, and includes parts intended to be incorporated into energy-related products covered by this Directive which are placed on the market and/or put into service as individual parts for end-users and of which the environmental performance can be assessed independently* (European Parliament, 2009)

[12] Concerning the "House", note that a low-energy house is considered. Similar considerations can be found in Blengini and Di Carlo (2010), where the contribution of the use phase of low-energy houses is

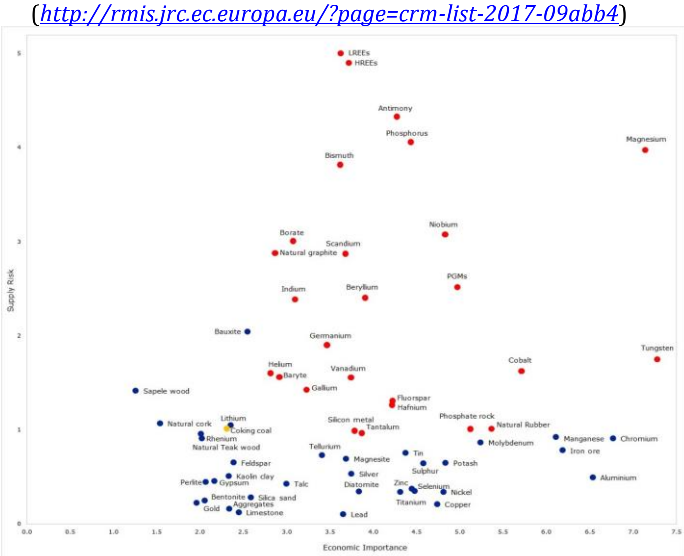
**BOX 3: Critical Raw Materials (CRMs)**

The revised list of CRMs for the EU was published by the EC in 2017 (EC, 2017a) and it includes 27 raw materials. Compared to the list published in 2014, there are 9 materials more. Note that all raw materials are considered important for the EU, even if they do not belong to this list since they are part of the manufacturing of products; hence, variation in flows of products, or trade policies or arising of new processes along the products' value change may have significant consequences in availability of raw materials. This is strictly connected to the need of increasing the recovery of all raw materials, especially those that are critical for the EU (EC, 2017a). Also for this reason, CRMs are considered as a priority in the CE Action Plan, which underling the importance of fostering their efficient use (EC, 2015).

It is to be noticed that in the above-mentioned documents, recycling is often mentioned as a practice to be improved. In fact, the recovery of materials from recycling, i.e. Secondary Raw Materials (SRMs), can participate in decreasing the demand of primary raw materials. However, an efficient use of materials also entail the extension of the lifetime of products; longer lasting products will translate in a delay in products available for recycling, i.e. in recovery raw materials. This aspects need to be captured and measured in order to better understand the benefits and /or drawbacks of extending the lifetime of products compared to their replacement (and materials recovery). The measurement of this aspect is particularly relevant for products with a high potential environmental impact or products embedding CRMs.

Indeed, for the characteristics of CRMs (i.e. supply and economic importance), the environmental and economic aspects should be both considered. Moreover, the evolution of the market and, consequently, of the flow of products, should complement the environmental and economic analysis to offer a broader and clearer picture of the system. Various studies in the literature quantitatively assess environmental aspects, economic aspects, and stocks and flows; however, very few studies combine these types of analyses.

Graphical representation of the Critical Raw Materials for the EU
(*http://rmis.jrc.ec.europa.eu/?page=crm-list-2017-09abb4*)




quite low compared to the contribution of the use phase of standard houses, which is more than 80% of the life-cycle energy use (considering a lifetime of 70 years)

In the scientific literature, studies assessing the impacts of reusing products already exist (Cooper and Gutowski, 2015; Hur et al., 2005; van Nes and Cramer, 2006; WRAP, 2010). Such studies mainly focus on the use phase of Electrical and Electronic Equipment (EEE). Moreover, the assessments of reuse of products often consider that the product does not change its function, i.e. it is repaired or remanufactured. However, there are examples of products that are refurbished/repurposed and then used in second-use applications, i.e. providing a function for which they were not originally conceived (section 1.3).

### 1.4.1 Currently available assessment tools

Among the available tools for assessing the environmental impacts of products in a life-cycle perspective, LCA is often adopted by the scientific community (section 1.1). It is a standardized methodology for the analysis of the environmental burden of products at all stages in their life cycle, "from-cradle-to-grave" (from the extraction of resources, through the production of materials, the use of the product, the possible reuse, recycling or final disposal), following the ISO 14040-14044 (ISO, 2006a).

In the framework of the environmental assessment of resource efficiency of Energy-Using Products, Ardente and Mathieux (2014b) developed an *ad hoc* method, namely "Resource Efficiency Assessment of Products" (REAPro). In the method, five indexes were used to identify relevant and efficient product's measures and to calculate the benefits of resource efficiency measures.

The detailed analysis of materials and process flows along their whole value-chain allows to obtain a more complete understanding of products' status (Nuss and Blengini, 2018). This also aligned with Ardente et al. (2017), De Meester et al. (2019) and Mancini et al. (2015), where it is underlined the need of combining together different assessment tools to fully capture the impacts of extending the lifetime of products, including also materials and resources, and to support decision-making. This is particularly relevant in case of CRMs embedded in products. In this perspective, Material Flow Analysis (MFA) (Brunner and Rechberger, 2004) can provide an overview of the effects of extending the lifetime of products on the embedded resources, giving an overview of the variation of stocks and flows of materials but also supporting the estimation of needed materials (e.g. primary raw materials) and the potential contribution of Secondary Raw Materials (SRMs) to the materials demand.

Overall, to capture both the impacts of extending the lifetime of products and its effects of resources, a life-cycle based and a multi-criteria method is needed. Available assessment tools focusing on different environmental aspects should be integrated in order to give a more complete overview of potential benefits of extending the lifetime of products.

## 1.5 Aim and outline of the thesis

The PhD research focused on the development of a methodological framework able to assess the environmental impacts of extending the lifetime of products. Consistent with the Introduction, extending the lifetime of products has a key role in boosting circularity of products, minimizing their environmental burdens and improving resource efficiency. The interest in extending the lifetime of products is attracting different actors of the products' value-chains, offering also new business opportunities. Despite several policy and scientific documents emphasize the relevance of different strategies of lifetime extension, existing indicators cannot quantitatively capture the potential benefits related to reuse. Moreover, the knowledge in this field is lacking; therefore, a collaboration with industrial stakeholders and policy makers involved in this emerging sector is needed to gather information and quantitative data to increase the knowledge and to quantify benefits of longer lasting products.

In this framework, the PhD focuses on the assessment of impacts of extending the lifetime of products through different strategies in a circular economy context. The methodological framework allows combining different assessment tools according to the characteristics of the assessed products group, hence taking into account the specificities of products. In fact, according to the product group and the available technologies (and/or existing researches), the lifetime of a product could be extended through its repair or remanufacturing, or repurposing and second-use, etc.

The environmental assessment tools adopted in the developed work are LCA, that allows to assess the impacts of products in a life-cycle perspective and it is a standardised methodology; the REAPro method, that allows to assess the resource efficiency of extending the lifetime of products; Material Flow Analysis (MFA), that allows to capture the effects of extending the lifetime of products through the variation of stocks and flows of products/materials. According to the characteristics of the products, such tools can be combined to provide a multi-criteria analysis and to offer a more complete overview and a more in-depth knowledge of the effects of lifetime extension of products.

LCA is combined with REAPro method. Bearing in mind that extending lifetime of products is also driven by socio-economic aspects, these two methods were enlarged to also include economic aspects. As a result, the *Environmental and Economic Assessment of Durability of Product* ("Pro-EnDurAncE") was extended (chapter 2). The environmental assessment of durability of products is illustrated in section 2.1.1, while the economic assessment, in section 2.1.2. This method is then applied to a product group for which the lifetime extension could have significant benefits in a life-cycle perspective since the contribution of the use phase is quite relevant (Figure 7 and Tukker et al., 2006): appliances in the housing sector. Durability requirements are already introduced by the EU legislation through Ecodesign measures for the vacuum cleaner (VC) product group (section 2.2). Due to the lack of specific data, a case-study canister VC was dismantled to obtain a Bill of Materials. These data were used to perform a detailed LCA (section 2.2.2). LCA results and the performed literature review, together with information gathered from stakeholders, represented the necessary background for the assessment of durability of the VC, according to the proposed method. Results of the environmental assessment are illustrated in section 2.2.3, while section 2.2.4 reports the results of the economic assessment.

As proved for the performed analysis, LCA is able to provide the necessary background information of the product/service under analysis, and this is particularly relevant in case of complex systems. The development of modular LCAs and the adoption of parameters makes the life-cycle model flexible to update according to available input data and to speed-up the LCAs of different products. The adoption of the same approach allows quick and consistent comparisons between environmental performances of different products/systems. Additionally, due to the fast development of the technology especially in specific sectors, the modularity of the LCA model allows to enlarge it, adding e.g. new materials and/or components. This modularity approach is therefore adopted to also model the life-cycle impacts in the automotive sector (chapter 3), which is one of the most relevant sector for the EU in terms of environmental impacts (Figure 7 and

Tukker et al., 2006). Thanks to the collaboration between *Politecnico di Torino* and *Centro Ricerche FIAT* (CRF), the impacts of different FCA vehicles were modelled through the development of a modular and parametrized model. Collaboration with CRF allowed gathering primary data on the vehicles composition, which is an important added value of LCA of vehicles. The added value of the developed modular LCA is represented by the direct link between the LCA model and the database of car manufacturers.

Still focusing on the automotive sector, environmental targets are driving the market towards e-mobility. Traction batteries represent the most important component in terms of costs but also of environment. In this framework, the extension of the lifetime of traction batteries can potentially have some benefits (chapter 4). Then, the above-mentioned assessment tools were used to assess the impacts of second-use of batteries in Europe. The same approach used for the environmental assessment of durability of products was used to develop an adapted LCA method to assess the environmental performance of extending the lifetime of traction batteries (section 4.2). Then, this method is applied to a case-study battery in two different applications (section 4.3). The lack of data for the modelling was addressed through the dismantling of a case-study battery; moreover, real load profiles were used to model the energy flows of the two systems in which the repurposed battery is supposed to be adopted.

Finally, the fast development of new products will inevitably affect the adoption of different new materials. This last aspect is particularly relevant for some Li-ion batteries (LIBs) chemistries as they embed some CRMs for the EU, e.g. cobalt. Therefore, the analysis of stocks and flows of products/materials was used to improve the knowledge of effects of second-use of batteries (chapter 5). Therefore, a dynamic MFA of the value chain of traction batteries after their use in Europe was developed (section 5.2) and the stocks and flows of batteries, of two embedded materials (cobalt and lithium) and the storage capacity of such batteries were estimated until 2030 (section 5.3). Parameters in the model allowed to assess different scenarios and the effects of extending the lifetime of batteries through their second-use in such stocks and flows.

Lessons learnt, recommendation and proposal for further work are described in chapter 6. Overall, the developed methodological framework contributes to the emerging field of resource efficiency and offers a framework to assess the environmental effects of extending the lifetime of products. Different stakeholders of the products' value-chain can adopt it to better understand the potential environmental benefits of extending the lifetime of products. The adoption of parameters in all the methodological components allows updating the analysis based on the availability of data. Multiple criteria are used to provide a more complete overview of the impacts (positive and/or negative) of complex systems. The developed methodological framework can be further extended to include multiple aspects and, if possible, to update models and data through a stricter collaboration with industrial stakeholders.

### 1.5.1 Publications (scientific journal)

The work during the PhD was developed under specific projects of the Joint Research Centre (JRC) of the European Commission. The main outcomes are published in scientific journal and JRC reports and are publicly available. Moreover, intermediate results were presented in some international conferences and discussed with experts of the field; so that, expert judgment and useful information for the work were collected and used to elaborate the methodological components illustrated in the following chapters.

Hereinafter, the most relevant publication and conference relevant for the presented topic are listed.


1. Bobba, S., Ardente, F., Mathieux, F., 2016. *Environmental and economic assessment of durability of energy-using products: Method and application to a case-study vacuum cleaner*. J. Clean. Prod. 137. doi:10.1016/j.jclepro.2016.07.093
2. Bobba, S., Mathieux, F., Ardente, F., Blengini, G.A., Cusenza, M.A., Podias, A., Pfrang, A., 2018. *Life Cycle Assessment of repurposed electric vehicle batteries: an adapted method based on modelling energy flows*. J. Energy Storage 19, 213–225. doi:10.1016/J.EST.2018.07.008
3. Bobba, S., Mathieux, F., Blengini, G.A., 2019. How will second-use of batteries affect stocks and flows in the EU? A model for traction Li-ion batteries. Resour. Conserv. Recycl. 145, 279–291. doi:10.1016/J.RESCONREC.2019.02.022
4. Cusenza, M.A., Di Persio, F., Bobba, S., Ardente, F., Cellura, M., 2019. *Energy and environmental assessment of a traction lithium-ion battery pack for plug-in hybrid electric vehicles*. J. Clean. Prod. 215, 634–649. doi:10.1016/j.jclepro.2019.01.056
5. Podias, A., Pfrang, A., Di Persio, F., Kriston, A., Bobba, S., Mathieux, F., Messagie, M., Boon-Brett, L., 2018. *Sustainability Assessment of Second Use Applications of Automotive Batteries: Ageing of Li-Ion Battery Cells in Automotive and Grid-Scale Applications*. World Electr. Veh. J. 9, 24. doi:10.3390/wevj9020024


## 1.5.2 Publications (reports)


1. Bobba, S., Ardente, F., Mathieux, F., 2015. *Technical Support for Environmental Footprinting, Material Efficiency in Product Policy and the European Platform on LCA - Durability assessment of vacuum cleaners. JCR Science and Policy Report*. JRC-EC (Joint Research Centre - European Commission). Available at publications.jrc.ec.europa.eu/repository/bitstream/JRC96942/lb-na-27512-en-n_.pdf. https://doi.org/10.2788/563222
2. Bobba, S., Podias, A., Di Persio, F., Messagie, M., Tecchio, P., Cusenza, M.A., Eynard, U., Mathieux, F., Pfrang, A., 2018. *Sustainability Assessment of Second Life Application of Automotive Batteries (SASLAB): JRC Exploratory Research (2016-2017): Final technical report: August 2018*. EUR 29321 EN, Publications Office of the European Union, Luxembourg, ISBN 978-92-79-92835-2; doi:10.2760/53624, JRC112543.
3. Mathieux, F., Ardente, F., Bobba, S., Nuss, P., Blengini, G.A., Alves Dias, P., Blagoeva, D., Torres De Matos, C., Wittmer, D., Pavel, C., Hamor, T., Saveyn, H., Gawlik, B., Orveillon, G., Huygens, D., Garbarino, E., Tzimas, E., Bouraoui, F., Solar, S., 2017. Critical raw materials and the circular economy - Background report. JRC-EC (Joint Research Centre - European Commission) Science-for-policy report, EUR 28832 EN, Publications Office of the European Union, Luxembourg. doi:10.2760/378123
4. Vidal-Legaz, B., Lucia Mancini, Gian Andrea Blengini, Claudiu Pavel, Alain Marmier, Darina Blagoeva, Cynthia E. L. Latunussa, et al. 2016. Raw Materials Scoreboard. JRC-EC (Joint Research Centre - European Commission). Available at https://publications.europa.eu/en/publication-detail/-/publication/1ee65e21-9ac4-11e6-868c-01aa75ed71a1/language-en. doi:10.2873/686373

### 1.5.3 Participation in conferences/workshop

1. S. Bobba, P. Tecchio, F. Mathieux,
F. Ardente, G.A. Blengini. *Sustainability assessment of 2nd life application*. Workshop "Second life of batteries : technical challenges and quality assessment" (Dusseldorf, 2019) (Speaker)
2. S. Bobba, F. Ardente, F. Mathieux, G. A. Blengini, F. Di Persio, A. Pfrang, M. Messagie, *Life Cycle Assessment (LCA) of repurposed traction Li-ion batteries in storage second-use applications*, ICBR International Congress for Battery Recycling 2018 (Berlin, 2018) (Speaker)
3. U. Eynard, S. Bobba, M. A. Cusenza, G. A. Blengini, *Lithium-ion batteries for electric vehicles: combining Environmental and Social Life Cycle Assessments*, Rete Italiana LCA (Messina, 2018) (Speaker)
4. F. Mathieux, A. Pfrang, S. Bobba, F. Di Persio, M. Messagie. *SASLAB - Sustainability Assessment of Second Life Application of Automotive Batteries*. Outreach workshop with policy DGs (Brussels, 05/06/2018) (Speaker)
5. M. Cusenza, S. Bobba, G.A. Blengini, M. Cellura, *Resource depletion of a Lithium ion battery cell technology*, SETAC Europe 28th Annual Meeting (Rome, 2018)
6. A. Pfrang, A. Podias, S. Bobba, F. Di Persio, M. Messagie, F. Mathieux, *Second Life Application of Automotive Li-Ion Batteries: Ageing During First and Second Use and Life Cycle Assessment*, Transport Research Arena (Vienna, 2018)
7. A. Podias, F. Di Persio, A. Pfrang, S. Bobba, P. Tecchio, F., Maarten Messagie. *SASLAB - Sustainability Assessment of Second Life Applications of Automotive Batteries*. JRC-EC Exploratory Research Workshop (Ispra 2018) (Speaker)
8. S. Bobba, F. Mathieux, A. Pfrang, G. A. Blengini, *Environmental assessment of potential second use of traction Li-ion batteries*, Circular economy perspectives for future end-of-life EV batteries (Brussels, 2017) (Speaker)
9. A. Podias, A. Pfrang, F. Di Persio, A. Kriston, S. Bobba, F. Mathieux, M. Messagie and L. Boon-Brett, *Sustainability Assessment of Second Life Application of Automotive Batteries: Ageing of Li-Ion Battery Cells in Automotive and Grid-scale Applications*, EVS30 Symposium (Stuttgart, 2017)
10. S. Bobba, G. A. Blengini, F. Di Persio, A. Podias, A. Pfrang, M. Messagie, F. Mathieux, *Second use of traction Li-ion batteries: an investigation of environmental performances based on material flow analysis*, ICBR International Congress for Battery Recycling 2017 (Lisbon, 2017) (Speaker)
11. A. Podias, A. Pfrang, F. Di Persio, A. Kriston, S. Bobba, F. Mathieux, M. Messagie and L. Boon-Brett, *Sustainability Assessment of Second Life Application of Automotive Batteries: Preliminary results on ageing of Li-ion cells in automotive applications and power grid support*, Advanced Battery Power 2017 conference (Aachen, 2017)
12. F. Ardente, P. Tecchio, S. Bobba, F. Mathieux, *Assessment of resource efficiency in a life cycle perspective: the case of reuse*, Rete Italiana LCA (Siena, 2017)
13. A. Podias, F. Di Persio, A. Pfrang, S. Bobba, P. Tecchio, F., Maarten Messagie. *SASLAB - Sustainability Assessment of Second Life Applications of Automotive Batteries*. JRC-EC Exploratory Research Workshop (Ispra 2017) (Speaker)
14. F. Di Persio, A. Pfrang, A. Podias, S. Bobba, F. Mathieux, M. Messagie, *Sustainability Assessment of Second Life Application of Automotive Batteries (SASLAB)*, ICBR International Congress for Battery Recycling 2016 (Antwerp, 2016)

15. F. Di Persio, A. Pfrang, A. Podias, S. Bobba, F. Mathieux, M. Messagie, *SASLAB - Sustainability Assessment of Second Life Application of Automotive Batteries*, Batteries 2020 workshop (Brussels, 2016) (Speaker)
16. Global Cleaner Production and Sustainable Consumption Conference 2015, 1-4/11/2015, Sitges (Spain). Speaker of the oral presentation about *Environmental and economic assessment of durability of vacuum cleaners* (Speaker)

# Environmental and Economic Assessment of Durability of Products. Application to the Energy-related Products (ErP)

In the proposed methodological framework, the first methodological components is represented by the Pro-EnDurAncE method *(Environmental and Economic Assessment of Durability of Product*. To develop such a method, the *Life Cycle Assessment* (LCA) and *Resource Efficiency Assessment of Products* (REAPro) methods are combined to assess the environmental performances of reusing products and are combined with an economic assessment to also capture the potential economic benefits related to the extension of lifetime of products.

In this chapter, the integration of these two assessment tools and their combination with economic aspects is described. Section 2.1 reports the detailed description of the Pro-EnDurAncE method. In particular, the environmental assessment of durability of products (described in section 2.1.1) represents a further development of the REAPro method, which aims at assessing the resource efficiency of products. Thanks to the information gathered during the work and the opinion of experts and stakeholders, a similar approach (section 2.1.2) was adopted to also assess the economic effects of extending the lifetime of products. In fact, both the literature and experts of the field recognised that economic aspects could strongly affect the adoption of longer lasting products by consumers and play a key role to incentivize reuse strategies.

Then, the developed method is applied to a case-study product. The identify product belongs to the energy-related products (ErPs), which are very important in terms on environmental impact among the “housing” products (Tukker et al., 2006). In particular, the quantitative assessment of both environmental and economic impacts of extending the lifetime focused on the VCs product group. It is also highlighted that for VCs, durability requirements have been introduced into Ecodesign measures, already (EC, 2008) (section 2.2). Conclusions of the study are reported in section 2.3.

Note that this chapter is mainly refer to (Bobba et al., 2015) and (Bobba et al., 2016a). The study is developed in the framework of a traineeship at the Joint research Centre (JRC) of the European Commission.

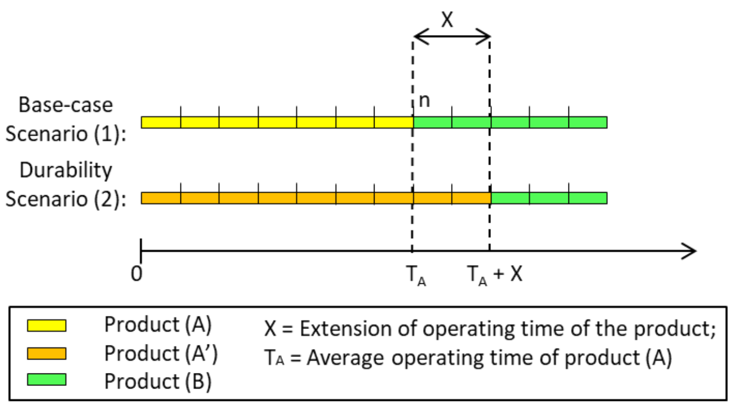
## 2.1 Environmental and economic assessment of durability of products (Pro-EnDurAncE)

To assess the environmental performances of products, the REAPro (Ardente and Mathieux, 2014b) is used as started point. Such a method was enlarged to include additional factors and also assess economic aspects. The detailed updates are reported in both (Bobba et al., 2015) and (Bobba et al., 2016a), while hereinafter significant aspects of the method are reported.

### 2.1.1 Method for the environmental assessment of durability of products

The environmental assessment of durability of products is based on the comparison of two different scenarios: a Base-Case Scenario and a Durability Scenario. In the Base-case scenario, a standard product (A), at the end of its operating life, is substituted by a new product (B), while in the Durability scenario the lifetime of a product with the same function is extended through repair operations (eventually).

Figure 8: Scenarios used for the environmental assessment of the durability of products




As defined by Ardente and Mathieux (2014a), the durability index ($D_n$) for a general impact category "n" is illustrated in Formula 1.

Formula 1:

$$D_n = \frac{\frac{P_{B,n}}{T_B} \cdot X + \frac{E_{B,n}}{T_B} \cdot X + \left(U_{B,n} - U_{A,n}\right) \cdot X - R_{A,n}}{P_{A,n} + U_{A,n} + E_{A,n}} \cdot 100 \quad [\%]$$

Where:

- $D_n$ = Durability index for the impact category "n" [%]
- $P_{B,n}$ = Environmental impact for category "n" for the production of product (B) (including the production of raw materials and manufacturing) [unit];
- $T_B$ = Average operating time of product (B) [hour];
- X = Extension of operating time of product (A) [hour];
- $E_{B,n}$ = Environmental impact for category "n" for the EoL treatments of product (B) [unit];
- $U_{B,n}$ = Environmental impact per unit of time for category "n" for the use of product (B) [unit/hour];
- $U_{A,n}$ = Environmental impact per unit of time for category "n" for the use of product A [unit/hour];
- $R_{A,n}$ = Environmental impact per unit of time for category "n" for additional treatments (e.g. repairing, refurbishment) necessary for the extension of operating time $T_A$ [unit];
- $P_{A,n}$ = Environmental impact for category "n" for the production of product (A) (including the production of raw materials and manufacturing) [unit];

- $E_{A,n}$ = Environmental impact for category "n" for the EoL treatments of product (A) [unit].

The environmental impact of the specific life-cycle stage is proportional to the lifetime extension (X).

This method was updated including also the contribution of auxiliary materials, of replacing product and of additional impacts of durable product to the life-cycle impact.

#### 2.1.1.1 The use phase (energy consumption, auxiliaries and maintenance)

Despite the huge impact related to the use phase (use-phase dominant product), some ErPs need auxiliaries' materials for accomplishing their duties. Hence, the environmental impact $U_n$ can be expressed as:

Formula 2:

$$U_n = AU_n + M_n + u_n$$

Where:

- $AU_n$ = Environmental impact for category "n" for the auxiliaries' materials consumption [unit/hour];
- $M_n$ = Environmental impact for category "n" for the maintenance [unit/hour];
- $u_n$ = Environmental impact for category "n" for the energy consumption during the use phase [unit/hour].

The functionality of product A' is the same as product (A). Therefore, the use of auxiliaries is supposed to be the same (i.e. $AU_A = AU_{A'}$). Similarly, for the impact of the maintenance ($M_A = M_{A'}$) and the energy consumption ($u_{A,n} = u_{A',n}$).

The environmental impact per unit of time for the use of product (B) can be expressed as a certain percentage (δ) of the environmental impact per unit of time for the use of product (A).

Formula 3:

$$u_B = \delta \cdot u_A$$

As discussed in Ardente & Mathieux (2014), it is always environmental convenient to prolong the lifetime of a product if $U_B > U_A$ [13]. Therefore, the analysis focuses to the case that the product (B) is more energy efficient than product (A), i.e. when:

$$0 < \delta < 1$$

#### 2.1.1.2 Replacing product

To take into account the potential development of technology and the changes of manufacturing processes, it is necessary to estimate the impacts for the production ($P_{B,n}$) of the

[13] Considering the technological progress, it is plausible to assume that the environmental impact for the use of product (B) would be lower than the environmental impact for the use of product. However, sometimes modern products can consume more energy due, for example, to additional functions implemented

replacing product (B). $P_B$ can be expressed in function of the impact for the manufacturing of product (A) taking into account a parameter ($\gamma$)[14]:

Formula 4:

$$\gamma = \frac{P_{B,n}}{P_{A,n}} \cdot, \gamma > 0$$

### 2.1.1.3 Additional impacts of durable product

The design of more durable products could imply additional burdens for example due to the use of additional/higher quality materials (Kostecki, 1998; Mora, 2007; AEA Energy & Environment 2009), and this variation could entail some additional impacts of durable products: longer design processes, development of innovative machineries, more tight testing, etc.[15]

Hence, impacts of durable products manufacturing ($P'_{A,n}$) can be expressed in function of the impact for the manufacturing of product (A) ($P_{A,n}$), as:

Formula 5:

$$P_{A\prime,n} = (1 + \alpha) \cdot P_{A,n} \qquad ; \qquad \alpha = \frac{(P_{A\prime,n} - P_{A,n})}{P_{A,n}} \qquad \alpha \geq 0$$

where:

- $P_{A\prime,n}$ = Environmental impact for category "n" to make product (A') more durable (including all the impact for the production of raw materials and manufacturing) [unit];
- $P_{A,n}$ = Environmental impact for category "n" to make product (A) more durable (including all the impact for the production of raw materials and manufacturing) [unit];
- $\alpha$ = Percentage representing the higher impact to make product (A) more durable [%].

For example, a value of ($\alpha$ = 10%) implies that 10% additional impacts are necessary to make base-case product more durable.

Taking into account all the previous considerations, the updated Durability Index is illustrated in the following formula.

---

[14] Values of $0 < \gamma < 1$ imply that impact to manufacture product (B) are lower than those of product (A) (e.g. due to dematerialization of the product); values $\gamma$>1 imply that impact to manufacture product (B) are higher than those of product (A) (e.g. due to increased complexity of the product and its electronic components)

[15] For example the preparatory study on VC identified as a possible option to increase the product lifetime by 50%, which is "likely to impinge on the need to improve the durability of the vacuum cleaner itself (i.e. increased weights of materials to strengthen items) (AEA Energy & Environment 2009). The study suggests that more materials and thus more environmental impacts are involved in the manufacturing of the more durable product. However, the report did not detail how this was modelled in the Ecoreport tool and how much additional materials have been accounted.

$$D_n = \frac{\frac{(\gamma - \alpha) \cdot P_{A,n}}{T_B} \cdot X + \left[\frac{E_{B,n}}{T_B} \cdot X + (E_A - E_{A'})\right] + \left(AU_{B,n} + M_{B,n} - AU_{A,n} - M_{A,n}\right) \cdot \mathrm{X} - (1 - \delta) \cdot u_{A,n} \cdot X - R_{A,n}}{P_{A,n} + U_{A,n} \cdot T_A + E_{A,n}} \cdot 100$$

Note that the impacts for repair ($R_A$) negatively affect the index, meaning that if repair operations occur to prolong the lifetime (for instance repairs or replacement of some components), then the more durable product can be less convenient from an environmental point of view. This confirms that reparability is a key aspect for the assessment of the durability of products, in line with the analysis of the relevant literature.

### 2.1.2 Method for the economic assessment of durability of products

The economic aspects related to acquisition, operating and maintenance are drivers for user decision regarding the substitution or, instead, their maintenance/repair to prolong their durability. So that the environmental assessment of durability is coupled with an economic assessment in order to enlarge the analysis and to have a more complete overview of durability.

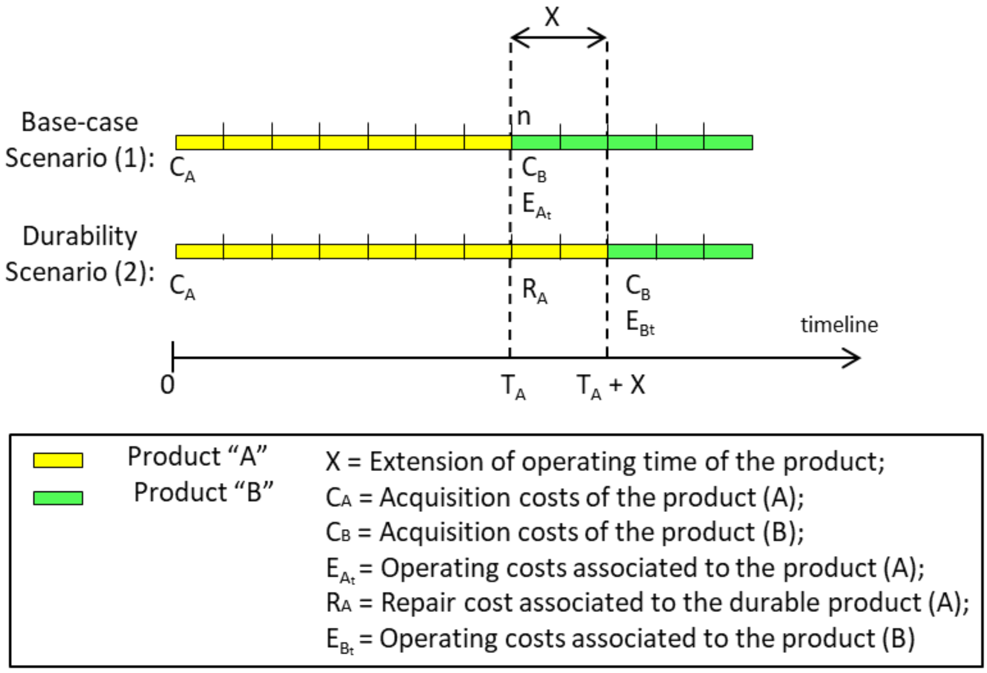
Also in case of economic assessment, the method is based on a similar approach to that applied for the environmental assessment, comparing two scenarios in which a standard product is compared with more durable products (Figure 9).

In the Base-case scenario, the base-case product (A) is replaced by a more energy-efficient product (B) after its average lifetime ($T_A$). The Durability scenarios implies that the product (A) lasts longer than in the previous scenario ($T_A + X$), and then it is replaced by a more energy-efficient product (B). In this scenario, additional expenditure compared to the Base-case scenario can occur, e.g. due to an extraordinary maintenance or repairing some components of the VC.

The expenditure occurring every year by the consumers for the operation, as well as the purchase price of product (B), the maintenance of both products ($M_{At}$ and $M_{Bt}$), and the *repair cost* ($R_A$) are discounted in order to evaluate their present value through a Present Value Factor (PVF)[16]. Moreover, the purchase price of the new product have to be distributed proportionally to the average operating time of product in order to allow the comparison of the two scenarios.

[16] The Present Value is the value of an expected cash flow stream determined as of the date of valuation, meaning discounted at the discount rate. The higher the discount rate, the lower the present value of the future cash flows.

Figure 9: Setting of scenarios to be compared in the assessment of durability




Formula 6: Base-case scenario:

$$C_{TOT,base\ case} = C_A + \sum_{t=1}^{T_A} [PVF_{t_B,i}(E_{A_t} + M_{A_t} + AU_{A_t})] + (PVF_{t,i}C_B) \cdot \frac{X}{T_B} + \sum_{t=1}^{X} [PVF_{t_B,i}(E_{B_t} + M_{B_t} + AU_{B_t})]$$

Where:

- PVF = Present Value Factor of the cash flow stream considered [-];
- i = Discount rate [%];
- t = Generic period of time for which the cost is calculated [y]
- $C_A$ = Acquisition costs of the product A [€];
- $T_A$ = Operating lifetime of product A [h];
- $E_{A_t}$ = Operating costs associated to the product (A) [€/y];
- $M_{A_t}$ =Maintenance costs associated to the product (A) [€/y];
- $AU_{A_t}$ = Auxiliaries components costs associated to the product (A) [€/y];
- $C_B$ = Acquisition costs of the product (B) [€];
- X = Extended lifetime [hours];
- $T_B$ = Operating lifetime of product (B) [h];
- $E_{B_t}$ = Operating costs associated to the product(B) [€];
- $M_{B_t}$ =Maintenance costs associated to the product (B) [€/y];
- $AU_{B_t}$ = Auxiliaries components costs associated to the product (B) [€/y];

Formula 7: Durability scenario:

$$C_{TOT,durable} = C_A + \sum_{t=1}^{T_A+X} [PVF_{t_A,i}(E_{A_t} + M_{A_t} + AU_{A_t})] + PVF_{t,i}(R_A)$$

Where:

- $C_A$ = Acquisition costs of the product A' [€];
- $T_A + X$ = Operating lifetime of the durable product A' [h];
- $E_{A_t}$ = Operating costs associated to the product (A) [€/y];
- $M_{A_t}$ =Maintenance costs associated to the product (A) [€/y];
- $AU_{A_t}$ = Auxiliaries components costs associated to the product (A) [€/y];
- $R_A$ = Repair cost associated to the durable product A' [€].

Comparing the two scenarios[17]:

Formula 8:

$$\Delta_{C_{TOT}} = C_{TOT,base\ case} - C_{TOT,\mathrm{durable}}$$

$$\Delta_{C_{TOT}} = \sum_{t=1}^{T_A}[PVF_{t_B,i}(E_{A_t} + M_{A_t} + A_{A_t})] + (PVF_{t,i}C_B)\cdot\frac{X}{T_B} + \sum_{t=1}^{X}[PVF_{t_B,i}(E_{B_t} + M_{B_t} + A_{B_t})] - \sum_{t=1}^{T_A+X}[PVF_{t_A,i}(E_{A_t} + M_{A_t} + A_{A_t})] - PVF_{t,i}(R_A)$$

$$\Delta_{C_{TOT}} = (PVF_{t,i}C_B)\cdot\frac{X}{T_B} + \sum_{t_B=1}^{X}[PVF_{t_B,i}(E_{t_B} + M_{t_B} + A_{t_B})] - \sum_{t_1=T_A+1}^{T_A+X}[PVF_{t_A,i}(E_{t_A} + M_{t_A} + A_{t_A})] - PVF_{t,i}(R_A)$$

Where:

- $\Delta C_{TOT}$ = difference between the life-cycle costs between the Durability scenario and the Base-case scenario.

Economic benefits occur when the difference between the life-cycle costs between the two scenarios is positive ($\Delta C_{TOT} > 0$), i.e. when the life-cycle costs of the Base-case Scenario are higher than the total costs of the Durability Scenario ($C_{TOT,base-case} > C_{TOT,durable}$). Note that the economic result is independent from the purchase price of the product (A) (as it is the same in both scenarios).

The cost related to the energy consumption depends on the product of the energy consumption ($e_n$) and the cost per [kWh] ($E_n$). Moreover, due to the higher energy efficiency of product (B), the energy consumption of the new product ($e_B$) can be assumed to be lower than of product A ($e_A$), and the costs of energy consumption of new product could be expressed as a percentage of the costs for the energy consumption of the old products:

Formula 9:

$$\frac{e_B}{e_A} = \delta\,,\ E_{t_B} = e_B\cdot E_t = \delta\cdot e_A\cdot E_t\,, \qquad 0 \le \delta$$

[17] $\sum_{k=1}^{n+m} a_k = \sum_{k=1}^{n} a_k + \sum_{k=n+1}^{n+m} a_k$

Due to lack of data on future cost trends, the purchase price and the maintenance costs of product (B) can be expressed as function of the product (A):

Formula 10:

$$C_B = (1 + \beta) \cdot C_A$$

Formula 11:

$$M_B = \rho \cdot M_A$$

Formula 12:

$$A_B = \sigma \cdot A_A$$

Therefore, the $\Delta C_{TOT}$ can be written as in Formula 13

Formula 13:

$$\Delta_{C_{TOT}} = (PVF_{t,i} C_B) \cdot \frac{X}{T_B} + \sum_{t=1}^{X} \{PVF_{t_B,i}[(\delta \cdot e_{A,t} \cdot E_t) + (\rho \cdot M_{A,t}) + (\sigma \cdot A_{A,t})]\} - \sum_{t_1=T_A+1}^{T_A+X} [PVF_{t_A,i}(e_{A,t} \cdot E_t + M_{A,t} + A_{A,t})] - PVF_{t,i}(R_A)$$

## 2.2 Environmental and economic assessment of durability of Vacuum Cleaners (VCs)

The Pro-EnDurAncE was tested to a case-study product group in the framework of "Technical support for Environmental Footprint, material efficiency in product policy and the European Platform on LCA" funded by DG Environment (AA JRC No 33446 – 2013-11 07.0307/ENV/2013/SI2.668694/A1).

Before reporting the 2.2 Environmental and economic assessment of durability of VCs and the obtained results, the main outcomes of the performed literature review of relevant aspects affecting the potential extension of lifetime of products is provided (section 2.2.1). To apply the Pro-EnDurAncE method, a performed LCA provided the necessary information. Section 2.2 shows the performed LCA of a case-study VC. Note that during the Life Cycle inventory phase, useful information also for the economic assessment were gathered from both the literature and the consulted stakeholders. Finally, sections 2.2.3 and 2.2.4 respectively report the environmental the economic assessment of the case-study VC.

### 2.2.1 The VC product group: a case-study

In the last years, there were some examples of durability aspects introduced into mandatory European policies concerning household appliances (e.g. Ecodesign implementing measures for some Energy related Products - ErP) since they were recognised *among those products "offering a high potential for cost-effective reduction of greenhouse gas emissions"* (European Parliament, 2009). Products that should be covered by the implementing measures of the Ecodesign Directive must respect three criteria:

1. large quantities of products placed on the EU market;
2. significant environmental impact related to energy consumption;
3. significant potential improvement without entailing excessive costs.

In 2006, the apparent consumption[18] of domestic VCs in EU25 was around 45 million, most of them imported from China (only about 14 million produced in EU-27, particularly in Germany, Italy and UK). 85% of the VCs on the market are canister VCs, while just 15% are uprights' (AEA, 2009). VCs sales growth rate between 2000 and 2005 was around 9% (trade data and PRODCOM statistics[19]), and most of the sold units were related to replacements, but during the crisis 2008-2009-2010 the sales drop to the level of around 2005, and afterwards started again to increase (AEA, 2009; EC, 2013). The main reasons for the increase of the sales of VCs between 2000 and 2005 may be related to the VC lifetime: the assumed average lifetime for domestic VCs is equal to 8 years, but it is expected to decline from 8 years in 2010 to 5 years in 2020 (AEA, 2009). In addition, the overall amount of electricity consumption associated to the operation of VCs is very high[20]: around 19 TWh per year in the EU-27 (about 25% due to non-domestic VCs, and 75% to domestic ones).
Therefore, VCs respect all these three criteria (AEA, 2009). Then, VC is one of the few product groups for which durability requirements have been introduced into Ecodesign measures (EC, 2008).

*Ecodesign requirements for the durability of VC have been enforced for the hose ("the hose, if any, shall be durable so that it is still useable after 40,000 oscillations under strain") and the operational motor lifetime ("the operational motor lifetime shall be greater than or equal to 500 h")* (EU, 2013a). These two components are in fact responsible for about 30% of the VC breakdowns (Table 1).

Table 1: Main reasons for vacuum cleaners breakdowns (AEA, 2009)

| **Reason for breakdown** | **Upright** | **Cylinder** |
|---|---|---|
| Split/broken hose | 21% | 25% |
| Suction | 19% | 15% |
| Motor | 16% | - |
| Broken casing | - | 11% |
| Power cable | - | 11% |

*Moreover, starting from 01/09/2017, new maximum energy consumption thresholds entered into force for the VCs* (EC, 2014a; EU, 2013a). *Therefore, it is expected that future generation of VCs entering the market will be more and more efficient* (Table 2). In parallel to improvement of energy efficiency, a development in technology is expected especially in adopting new materials (e.g. to manufacture lighter VC), in decreasing the use of auxiliaries (e.g. bag-less VC) or in more complex

[18] The apparent consumption of VCs is given by the production and the imports of VC minus the exports of VCs
[19] http://www.eea.europa.eu/data-and-maps/data/external/prodcom-database-eurostat
[20] According to the available information of both the Preparatory study and the Impact assessment report, the environmental impact assessment of the EU-Stock 2005 highlights an electricity consumption of 3.7 TWh; moreover, in 2010 the EU-27 stock is of about 200 million units

PCB (as experienced for other EEE, more recent and advanced VC will require more complex PCB).

Table 2: Energy efficiency classes of VC before and after 2017

| Energy Efficiency Class | Annual energy consumption (AE) [kWh/y] | |
|---|---|---|
| | **From 1 September 2014** | **From 1 September 2017** |
| A+++ | n/a | AE ≤ 10,00 |
| A++ | n/a | 10,00 < AE ≤ 16,00 |
| A+ | n/a | 16,00 < AE ≤ 22,00 |
| A | AE ≤ 28,00 | 22,00 < AE ≤ 28,00 |
| B | 28,00 < AE ≤ 34,00 | 28,00 < AE ≤ 34,00 |
| C | 34,00 < AE ≤ 40,00 | 34,00 < AE ≤ 40,00 |
| D | 40,00 < AE ≤ 4616,00 | AE > 40,00 |
| E | 46,00 < AE ≤ 52,00 | n/a |
| F | 52,00 < AE ≤ 58,00 | n/a |
| G | AE > 58,00 | n/a |

The performed literature review aimed at gathering information on both durability aspects of VC and LCA information (Table 3), following different criteria:

- considerations about the durability of EEE, and VCs in particular (quantitative and qualitative dissertations);
- considerations about the lifetime of EEE and its dependency to specific aspects;
- some specific LCA steps, with reference to the Life Cycle Inventory (LCI) and the Life Cycle Impact Assessment (LCIA) of EEE, and VCs in particular;
- economic considerations associated to the life-cycle of EEE and/or to the durability issue;
- considerations about the consumer's role affecting durability (for instance time spent cleaning, obsolescence of products, maintenance and repair operations);
- considerations about the existence and the applicability of standards and test methods about VCs.

In general, VCs can be considered as a product with a balanced relation between its product value and its environmental impact, thus an eco-efficient[21] product category (Barba-Gutiérrez et al. 2008; Kobayashi et al. 2005). The available LCA point out that VC are "use-phase dominant" products (Abele et al. 2005; Hur et al. 2005; Kobayashi et al. 2005; Kemna et al. 2005; van Nes & Cramer 2006; Kota & Chakrabarti 2007; AEA Energy & Environment 2009; Gandy et al. 2012;), stressing the relevance of the energy consumption during operation. The evaluation of energy use and the corresponding environmental impact should consider different aspects simultaneously: the number of households[22], the dwelling sizes[23] and the power consumption of VCs (AEA, 2009; EC, 2013). Often consumer choice is based on the latter parameter: the higher is the input power,

[21] Eco-efficiency is defined as "product value per unit of environmental impact" (Kobayashi et al., 2005)

[22] The number of households is supposed to increase around 1-1.5% per year (EC, 2013)

[23] The dwelling size is supposed to be incremented by 20% in the period 2000-2020 (EC, 2013)

the higher seems to be the performance of the VCs. Mainly for that reason, in the last decades the input power range of VCs increased. On the contrary, "there is no correlation between input power and cleaning performance" (AEA, 2009). An in-depth knowledge of factors affecting the lifetime of VCs is fundamental for assessing the environmental impact and the potential savings associated to the life-cycle of VCs.

It is also noticed that manufacturing is always relevant, especially due to the impacts of motor and PCB (Abele et al., 2005; Barba-Gutiérrez et al., 2008; Gandy et al., 2012; Hur et al., 2005; Kobayashi et al., 2005; Kota and Chakrabarti, 2007; van Nes and Cramer, 2006; WRAP, 2010). Taking into account the potential improvements on these products related to the technologic development, several options should been explored in order to evaluate the potential benefits related to the extension of the VC lifetime and to the employment of less (for instance bag-less VC) or new materials (AEA Energy & Environment 2009; Kobayashi et al. 2005).

Moreover, some studies assume that the economic value (i.e. the VC price) will increase in the next decades, and that there is the possibility that this could be related also to higher environmental impacts, occurring especially in the manufacturing phase. More focus should be addressed to the reduction of the environmental impacts in order to encourage the use of more environmentally-friendly products even when more expensive (Barba-Gutiérrez et al., 2008; Hur et al., 2005; Kobayashi et al., 2005).

Table 3: Literature review on studies dealing with relevant aspects of durability of products

| N° | Authors | Durability | Lifetime indications | Environmental aspects | | Economic aspects | Consumer role |
|---|---|---|---|---|---|---|---|
| | | | | LCI | LCIA | | |
| 1 | (Rose, 2000) | X | X | - | - | - | X |
| 2 | (Ernzer and Birkhofer, 2003) | - | X | - | X | - | X |
| 3 | (Horie 2004); | X | X | X | X | X | X |
| 4 | (Abele et al., 2005) | - | X | - | X | X | X |
| 5 | (Cooper, 2005) | G.C. | X | - | G.C. | G.C. | G.C. |
| 6 | Hur et al. 2005) | - | - | G.C. | X | - | - |
| 7 | (Kemna et al., 2005) | X | X | X | X | X | X |
| 8 | (Kobayashi et al., 2005) | - | X | - | X | - | X |
| 9 | (Allenby, 2006) | X | X | X (EoL) | - | X | X |
| 10 | van Nes & Cramer 2006) | G.C. | G.C. | - | - | G.C. | G.C. |
| 11 | Kota and Chakrabarti (2007 | - | - | - | X | - | - |
| 12 | (Barba-Gutiérrez et al., 2008) | - | - | X | X | X | X |
| 13 | (AEA, 2009) | X | X | X | X | X | X |
| 14 | (Wong, 2009) | X | X | X | X | X | - |
| 15 | (Boustani et al., 2010) | - | X | X | X | X | - |
| 16 | (Cooper, 2010) | X | X | - | - | X | X |
| 17 | (Murakami et al., 2010) | X | X | - | - | - | X |
| 18 | (Maurer, 2010) | X | X | - | - | - | X |
| 19 | (WRAP, 2010) | - | G.C. | - | - | - | G.C. |

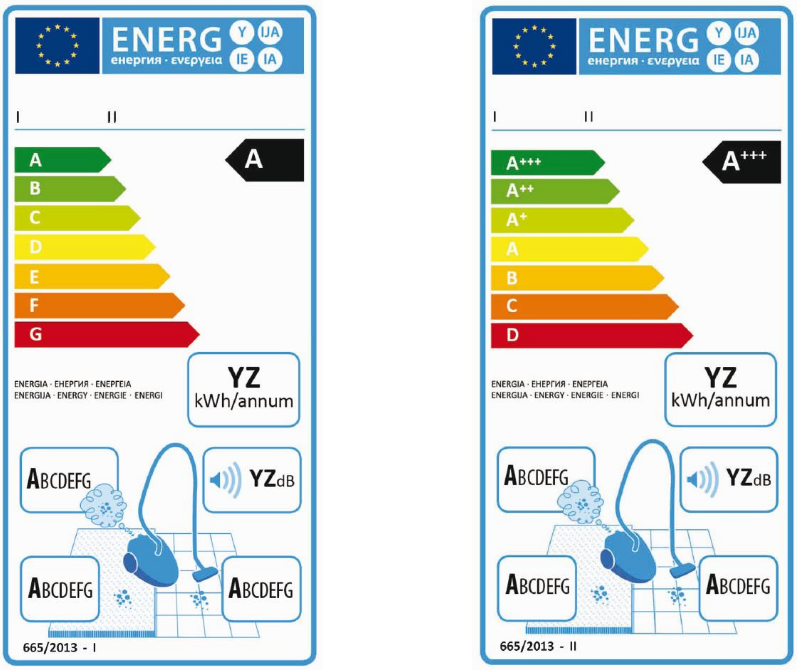
| | | | | | | | |
|---|---|---|---|---|---|---|---|
| 20 | (DEFRA, 2011) | G.C. | X | - | - | G.C. | G.C. |
| 21 | (WRAP, 2011) | X | X | - | - | X | X |
| 22 | (Huisman et al., 2012) | X | X | - | - | - | - |
| 23 | (BIOis 2013d) | X | X | - | - | X | X |
| 24 | (Monier et al., 2013) | - | X | - | - | - | - |
| 25 | (Tasaki et al., 2013) | X | X | - | - | - | X |
| 26 | WRAP 2013 | - | - | X (packaging) | - | - | - |
| 27 | (Boulos et al., 2014) | G.C. | X | - | - | G.C. | G.C. |
| 28 | (Sam et al., 2014) | G.C. | - | G.C. | G.C. | - | - |
| 29 | (TNS, 2014) | X | - | - | - | X | X |

G.C.: General Consideration, it means that the study does not refer specifically to VCs but give some information about ErPs

X: it means that the information within the study refer to VCs

Doubtless, the durability of VCs (and of EEEs in general) is strictly influenced by several factors such as consumers behaviours, reparability and reusability and existence of testing methods for assessing the longer lasting products performances (Boulos et al. 2014; BIO Intelligence Service 2013d; BIO Intelligence Service 2013b; BIO Intelligence Service 2013c; Abele et al. 2005). Hence, both industry and consumer have a key role for many of these aspects. the possibility of extending the lifetime of VC through a better design and its repair can be further incentivized through, for instance the easier replacement of specific components, an ease disassemblability, the availability of information for repair operation, the availability of both skills and spare parts but also the improvement of users awareness. An existing measure related to this last aspect is the "indication by labelling and standard product information of the consumption of energy and other resources by energy-related products" (EU, 2010), that allows end-users to choose more efficient products (Figure 10).

Figure 10: Energy label for vacuum cleaners placed on the market before (on the left) and after 1 September 2017 (on the right)



### 2.2.2 Life Cycle Assessment of a canister vacuum cleaner

The necessary background of the durability assessment as illustrated in section 2.1 is the LCA of a VCs. The relevant information of the performed LCA of a canister VCs are reported in this section (see (Bobba et al., 2015) for a more detailed description).

#### 2.2.2.1 Goal and scope

The overall aim of this LCA is to assess the potential environmental of an average canister VCs as it is the most representative VC type of the European market. The impact categories selected for this study are ILCD/PEF recommendation (EC - JRC, 2011)[24].

The functional unit of the study is a packaged canister VC (5.721 kg of mass for the VC and 1.26 kg for the packaging) with an operating lifetime of 10 years (corresponding to 50 hours per years of time spent vacuuming).

The system boundaries include the following phases:

- manufacturing phase (including transports for raw materials and the final delivery of the product);
- use phase, including consumption of electricity during the operation, use of auxiliaries components (periodical changes of dust-bags) and maintenance (periodical filters replacements);
- EoL, including transports and impact of waste treatment in a WEEE recycling plant.

The LCA of the case-study product is performed through GaBi6 software and the used database is PE database for almost all the process units. Though, in some cases, due to lack of information, Ecoinvent v2.2, DKI/ECI[25] and PE/FEFCO[26] databases are used (ANNEX1).

Data used for the model realization refer to a dismantled canister VC: all the components have been weighted and categorized by material (thanks to the brands). Where the information was not available, data are based on literature.

#### 2.2.2.2 Life Cycle Inventory (LCI)

The Bill of Material necessary to assess the impacts of a canister VC was derived from the dismantling of a canister VC with a mass of 5.721 kg (Figure 11). During the dismantling, each part was measured and the correspondent material identified. When necessary, exploded diagram of VC parts and information available on online manuals[27] were used to complement primary data.

---

[24] The land use impact category has been excluded (due to limited life cycle inventory data) and the Resource Depletion impacts have been subdivided into *Abiotic Depletion Potential, mineral resources[24]* and *Primary energy from non-renewable resources (net cal. value)*.

[25] German Copper Institute, http://www.kupferinstitut.de/

[26] European Database for Corrugated Board Life Cycle Studies, (FEFCO, 2011)

[27] http://www.manualowl.com/m/Hoover/S3332/Manual/380541 (accessed March 2015), http://www.manualowl.com/m/Hoover/S3332/Manual/182503?page=1 (accessed March 2015)

The components of the dismantled VC are illustrated in Table 4 while Table 5 depicts the detailed LCI and the main sources adopted to complement primary data. According Since the critical parts for durability requirements for VC are hose and operational motor, these parts have been investigated in detail.

Concerning the energy consumption for the manufacturing of VCs, information in the literature are lacking. Hence, studies on Electrical and Electronic Equipment were consulted and 10 [kWh] of electricity were assumed for the assembly of the VC (Boustani et al., 2010; Horie, 2004; Olivetti et al., 2012; Wong, 2009; WRAP, 2010).

An extensive literature review was performed to assume a suitable lifetime for the assessment. According to several sources (Table 6), the expected lifetime of VCs is ranging between 5 and 9 years, but some sources point out longer periods (e.g. some consumers' web-sites[28] or manufacturers[29]). Both the energy label and the Ecodesign implementing measures for VC assume 50 one-hour cleaning tasks per year; this means that, assuming 500 hours of lifetime of the motor, the VC can last about 10 years. Due to the relevance of economic aspects on the lifetime of products (section 1.2), the warranty offered by VCs' manufacturers and the availability of spare parts were also investigated. Depending on the manufacturer ad on the VC components, the years' warranty can assume different values, for instance 7 - 10 years on the motor and body casing[30], 5 years[31] or 2 years[32] for the VCs and its components.
For the analysis, it is assumed a lifetime of 10 years.

To model the energy consumption during the VC use phase, it was observed that the majority of VCs currently put into the market belongs mainly to the European energy class 'A' and only a small amount to class 'B'. Hence, the VC is supposed to belong to the energy class A in the energy labelling starting from September 2017 (EU, 2013b), i.e. a yearly energy consumption of 25 [kWh/y].

Concerning the maintenance of VC, the ordinary maintenance mainly entails the substitution and the cleaning of filters, whereas the extraordinary maintenance includes those operations which are not constantly carried out and deals with some specific components failure (Bobba et al., 2015). The hypothesis about ordinary maintenance for the present LCA is the usage of 1 set of filters per year, while the extraordinary maintenance is not considered in this specific LCA. In the modelling, also the potential adoption of auxiliary materials in included, as the case-study product is a bagged VC. According to the performed literature review, the frequency of dust-bags replacement is assumed equal to 7 bags per year[33].

---

[28] the life expectancy raise until 12 years

[29] Some manufacturers tested their vacuum cleaners to last 1,000 operating hours or an average of 20 years residential use and prove that the motor achieved between 800 and 950 hours and, if used consistently to the manual instruction, it can last up 1,200 hours (http://www.which.co.uk/news/2009/10/miele-ads-banned-for-vacuum-cleaner-claims-186889/ (accessed march 2015))

[30] http://www.achooallergy.com/miele.asp (accessed march 2015)

[31] http://www.dyson.com/vacuums/browsetherange.aspx (accessed march 2015)

[32] http://www.electrolux.com.au/Global-pages/Page-Footer-Menu/Top/Changes-to-Consumer-Law/ (accessed march 2015)

[33] This is the average value between those available in literature

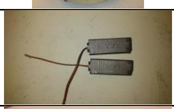
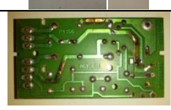
The modelling of the EoL was based on the EU Directive (2012/19/EC) (EU, 2012) regulating the WEEE recycling process and the minimum recycling/recovery rates for different fractions (separated before recycling/recovery)[34]. Waste produced during the use phase refer to the use of auxiliaries' components (i.e. dust-bags) and ordinary maintenance (i.e. filters). Also extraordinary maintenance produces wastage, but this amount can be considered as negligible (AEA, 2009; EC, 2013).

The environmental impacts of transport of waste and of shredding operations are included in the assessment. More in detail, transports, energy and chemicals for the processes are modelled based on the inventory provided by Huysman et al. (2015), whereas the recycling and the recovery rate of specific materials are assumed based on the IEC/TR 62635 (IEC, 2012). For the detailed LCI refer to (Bobba et al., 2015).

Figure 11: Case-study canister vacuum cleaner

[34] The WEEE Directive specifies that, starting from 15 August 2015, the minimum rate for small household appliances (SHA) recovery has to be 75% while at least 55% of SHA shall be prepared for re-use and recycled (2002/96/EC).

Table 4: Components of the dismantled canister vacuum cleaner

| N | Component | Description | Sub-category | LCA model category | N | Component | Description | Sub-category | LCA model category |
|---|---|---|---|---|---|---|---|---|---|
| 1 |  | Hose terminal | Hose | Hose | 26 |  | Pedal | Canister case | “Other components” |
| 2 |  | Hose handle | Hose | Hose | 27 |  | Pedal spring | Canister case | “Other components” |
| 3 |  | Tube extension | Hose | Hose | 28 |  | release lid | Canister case | “Other components” |
| 4 |  | Tube extension | Hose | Hose | 29 |  | Grip-handle | Canister case | “Other components” |
| 5 |  | Seal | Hose | Hose | 30 |  | Canister case, lower part | Canister case | “Other components” |
| 6 |  | Hose | Hose | Hose | 31 |  | Hose Air Regulator | Canister case | “Other components” |
| 7 |  | Connector | Motor | Motor | 32 |  | Canister case, upper part | Canister case | “Other components” |
| 8 |  | Fan housing | Motor | Motor | 33 |  | Padding indicator | Canister case | “Other components” |
| 9 |  | Seal | Motor | Motor | 34 |  | Filter | Filter | “Other components” |
| 10 |  | Stator assy | Motor | Motor | 35 |  | Filter | Filter | “Other components” |
| 11 |  | Fan | Motor | Motor | 36 |  | Filter | Filter | “Other components” |
| 12 |  | Carbon brushes | Motor | Motor | 37 |  | Nozzle | Nozzle | “Other components” |
| 13 |  | Rotor | Motor | Motor | 38 |  | Plastic components | Nozzle | “Other components” |
| 14 |  | Connectors | Motor | Motor | 39 |  | Wheels | Nozzle | “Other components” |
| 15 |  | Seal | Motor | Motor | 40 |  | Wheels axle | Nozzle | “Other components” |
| 16 |  | Washer | Motor | Motor | 41 |  | Nuts and springs | Nozzle | “Other components” |
| 17 |  | Motor cover | Motor | Motor | 42 |  | PCB | PCB | “Other components” |

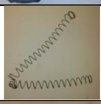
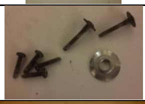
| | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|
| 18 |  | Guard motor | Motor | Motor | 43 |  | PCB | PCB | “Other components” |
| 19 |  | Springs | Motor | Motor | 44 |  | Power cord | Power cord reel | “Other components” |
| 20 |  | Nuts | | | 45 |  | Body | Power cord reel | “Other components” |
| 21 |  | Seal | Motor | Motor | 46 |  | Spring and Keeper | Power cord reel | “Other components” |
| 22 |  | Cable parts | Cables | “Other components” | 47 |  | Contact | Power cord reel | “Other components” |
| 23 |  | Cable parts | Cables | “Other components” | 48 |  | Seal | Power cord reel | “Other components” |
| 24 |  | Power button | Canister case | “Other components” | 49 |  | Canister wheels | Wheels | “Other components” |
| 25 |  | Canister case, middle part | Canister case | “Other components” | 50 |  | Canister wheel | Wheels | “Other components” |

Table 5: Bill of materials of the case study vacuum cleaner

| Component | Material | Quantity [g] | Note |
|---|---|---|---|
| Motor | Al | 7 | Additional details on the composition of these components derive from:<br>- motor: (De Almeida et al., 2013, 2008; European Alliance, 2011; Horie, 2004; Olivetti et al., 2012; Wong, 2009)<br>- bulk moulding compound – glass fibre (BMC-GF): MATBASE (2004), Prospector (2015) |
| | BMC-GF | 133 | |
| | brass | 99 | |
| | copper | 16 | |
| | graphite | 267 | |
| | others | 24 | |
| | PE | 158 | |
| | PP | 259 | |
| | rubber | 13 | |
| | steel | 885 | |
| Hose | ABS | 461 | |
| | PE-HD | 214 | |
| | PP | 18 | |
| | rubber | 3 | |
| Canister case | ABS | 2,004 | |
| | others | 8 | |
| | POM | 42 | |
| | rubber | 2 | |
| | steel | 4 | |
| Cable | brass | 2 | Details about the plastics composition of cord are derived from Baitz et al. (2004) |
| | copper | 7 | |
| | PE | 15 | |
| | PVC | 137 | |
| Cord reel assembly | ABS | 2 | |
| | brass | 89 | |
| | copper | 142 | |
| | PE | 21 | |
| | PVC | 52 | |

| | | | |
|---|---|---|---|
| | rubber | 4 | |
| | steel | 194 | |
| Dust bag | paper | 40 | Additional information, particularly about the amount of dust-bags used per year are derived from AEA (2009), Abele et al. (2005) and Kemna et al. (2005). |
| Filter | PE | 17 | Additional information, particularly about the amount of filters used per year are derived from Accumulair.com (2015), AchooAllergy.com (2015), AEA (2009), Dyson Company (2015a, 2015b) and Miele (2015) |
| Nozzle | ABS | 47 | |
| | PE | 20 | |
| | PP | 224 | |
| | steel | 19 | |
| PCB | PCB | 12 | Additional information, particularly about the amount of filters used per year are derived from the PE database (PE Europe GMBH, 2011) and Shenzhen Longood Electronics CO., (2015) |
| | steel | 14 | |
| | PP | 209 | |
| | ABS | 47 | |
| Wheel | PE | 20 | |
| Packaging | Cardboard | 1,100 | Packaging was modelled based on information derived from AEA (2009), Philips (2016), Suzhou KVC Electric Co. Ltd (2016) and WRAP (2013) |
| | LDPE | 60 | |
| | Paper (Manual) | 100 | |
| TOT | | 6,981 | |

Table 6: Considerations about vacuum cleaners lifetime in literature

| **Product** | **Lifetime** | **Note** | **Source** | **Year** |
|---|---|---|---|---|
| Vacuum cleaner | 5-7 years | "5-7 years lifespan. Clean the filters after every few uses to make sure nothing is blocking the nozzle, hose or entry to the bag." | http://classiccleaners.net/2013/09/03/whats-the-life-expectancy-of-these-5-appliances-part-1/ (accessed March 2015) | 2013 |
| Vacuum cleaner | 8 years | | "Okala Practitioner: Integrating Ecological Design" | 2013 |
| Vacuum cleaner | 5-7 years | "After every few uses, clean the filters to make sure nothing is blocking the nozzle, hose or entry to the bag or the dirt container." | (Johnston, 2013) | 2013 |
| Vacuum (canister, stick, upright, handheld) | 6-12 years (average = 9 years) | "Some models of vacuum cleaners have been demonstrated to have life spans in excess of 12 years." | Canadian Electrical Stewardship Association (CESA) | 2011 |
| Vacuum cleaner | 5-7 years | | Department for Environment, Food and Rural Affairs by brook Lyndhurst | 2011 |
| Vacuum cleaner | 550 hours | "Most consumers replace a vacuum cleaner due to a decrease in the cleaning performance of the product over time but only rarely because the motor is broken." | The European Association for the Co-ordination of Consumer Representation in Standardisation (ANEC) & the European Consumers' Organisation (BEUC) | 2010 |
| Vacuum cleaner | more than 1,200 hours | "Miele said its vacuum cleaner motors achieved between 800 and 950 hours on the highest setting, which was 300-450 hours above standard test regulations, and when the S7 was used on variable power settings, as recommended in the instruction manual, the motor lasted up to 1,200 hours" | http://www.which.co.uk/news/2009/10/miele-ads-banned-for-vacuum-cleaner-claims-186889/ (accessed March 2015) | 2009 |

| | | | | |
|---|---|---|---|---|
| Vacuum cleaner | 7 years | | A Practical Method for Quantifying Eco-efficiency Using Eco-design Support Tools | 2005 |
| Vacuum cleaner | up to 10 or 12 years | | http://www.vacuumcleanerrepair.org / (accessed March 2015) | n.a. |
| Vacuum cleaner | 15-20 years | | (Lee, 2015) | n.a. |
| Vacuum cleaner | more than 1,000 operating hours or 20 years | | http://www.achooallergy.com/miele-callisto-s5280-vacuum-cleaners.asp (accessed March 2015) | n.a. |

### 2.2.2.3 Life Cycle Impact Assessment (LCIA)

The life-cycle environmental impacts of the canister VC are shown in Table 7; Figure 12 reports the percentage contribution of life cycle phases.

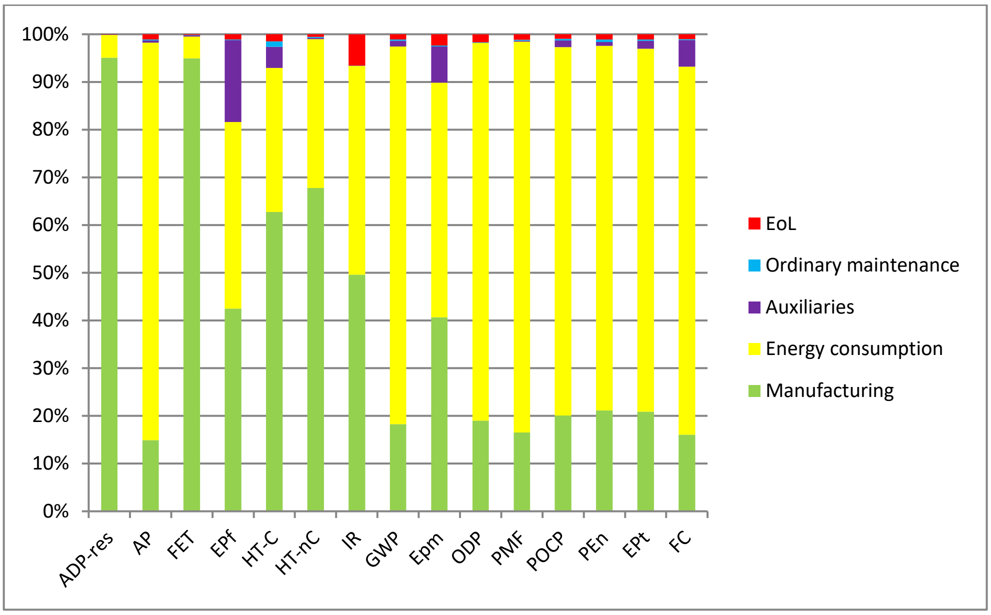
The major contributions in terms of environmental impacts are associated to use and manufacturing phases depending on the impact category. For the impact categories more related to the energy consumption (Acidification, GWP, Ozone Depletion and Particulate Matter) the energy consumption contribution is always higher than 79%. In any case, except for Ecotoxicity for aquatic fresh water and Abiotic Depletion, the use phase cannot be considered as negligible (less than 5%).
The manufacturing phase is particularly relevant (more than 95%) for Ecotoxicity for aquatic fresh water and Abiotic Depletion; in any case, it is never lowers than 14.5% for all the fifteen impact categories.
Overall, auxiliaries' components and ordinary maintenance have an environmental impact higher than 15% only for Freshwater eutrophication (6.55E-05 $kg_{Peq}$). The EoL phase contributes for less than 5% for almost all the impact categories; exception is represented by Ionising Radiation. Note that the system boundaries of the study have been set to include only the pre-processing of the VC waste, and not including the further treatments of recyclable/recoverable materials, nor the potential environmental credits due to the energy and material recovery.

Table 7: Life Cycle Impact Assessment of the canister vacuum cleaner

| Impact category | Unit of measure | Vacuum Cleaner | Manufacturing | Use phase | | | EoL |
|---|---|---|---|---|---|---|---|
| | | | | Auxiliaries components | Ordinary maintenance and dust-bags | Energy consumption | |
| ADP-res | [kg $Sb_{Equiv.}$] | 1,27E-03 | 1,21E-03 | 9,76E-07 | 1,80E-07 | 6,09E-05 | 9.16E-07 |
| AP | [Mole of $H+_{eq.}$] | 8,04E-01 | 1,20E-01 | 4,06E-03 | 1,60E-03 | 6,70E-01 | 8.64E-03 |
| FET | [CTUe] | 7,78E+01 | 7,39E+01 | 2,21E-01 | 5,09E-02 | 3,52E+00 | 1.22E-01 |
| EPf | [kg $P_{eq}$] | 3,79E-04 | 1,61E-04 | 6,50E-05 | 5,42E-07 | 1,48E-04 | 4.17E-06 |
| HT-C | [CTUh] | 3,37E-07 | 2,11E-07 | 1,48E-08 | 3,92E-09 | 1,02E-07 | 4.99E-09 |
| HT-nC | [CTUh] | 9,18E-06 | 6,22E-06 | 2,70E-08 | 1,79E-08 | 2,86E-06 | 4.92E-08 |
| IR | [kg $U235_{eq}$] | 5,59E+01 | 2,77E+01 | -1,27E-02 | 3,83E-02 | 2,44E+01 | 3.68E+00 |
| GWP | [kg $CO2_{Equiv.}$] | 1,49E+02 | 2,72E+01 | 1,80E+00 | 4,24E-01 | 1,18E+02 | 1.59E+00 |
| Epm | [kg $N_{Equiv.}$] | 1,06E-02 | 4,29E-03 | 8,04E-04 | 1,69E-05 | 5,19E-03 | 2.53E-04 |
| ODP | [kg CFC-$11_{eq}$] | 1,11E-07 | 2,11E-08 | -5,35E-11 | 1,39E-10 | 8,78E-08 | 1.82E-09 |

| Impact category | Unit of measure | Vacuum Cleaner | Manufacturing | Use phase | | | EoL |
|---|---|---|---|---|---|---|---|
| | | | | Auxiliaries components | Ordinary maintenance and dust-bags | Energy consumption | |
| PMF | [kg PM2,5$_{Equiv.}$] | 4,94E-02 | 8,17E-03 | 1,26E-04 | 8,75E-05 | 4,04E-02 | 5.71E-04 |
| POCP | [kg NMVOC] | 3,18E-01 | 6,39E-02 | 4,56E-03 | 1,03E-03 | 2,46E-01 | 3.10E-03 |
| PEn | [MJ] | 2,76E+03 | 5,84E+02 | 2,27E+01 | 1,46E+01 | 2,11E+03 | 3.01E+01 |
| EPt | [Mole of N $_{eq.}$] | 1,13E+00 | 2,37E-01 | 1,88E-02 | 2,74E-03 | 8,62E-01 | 1.25E-02 |
| FC | [UBP] | 1,60E+02 | 2,57E+01 | 8,94E+00 | 3,10E-01 | 1,24E+02 | 1.66E+00 |

Figure 12: Percentage contribution to the overall impact of the canister vacuum cleaner life-cycle phases




## 2.2.2.4 Life Cycle Interpretation

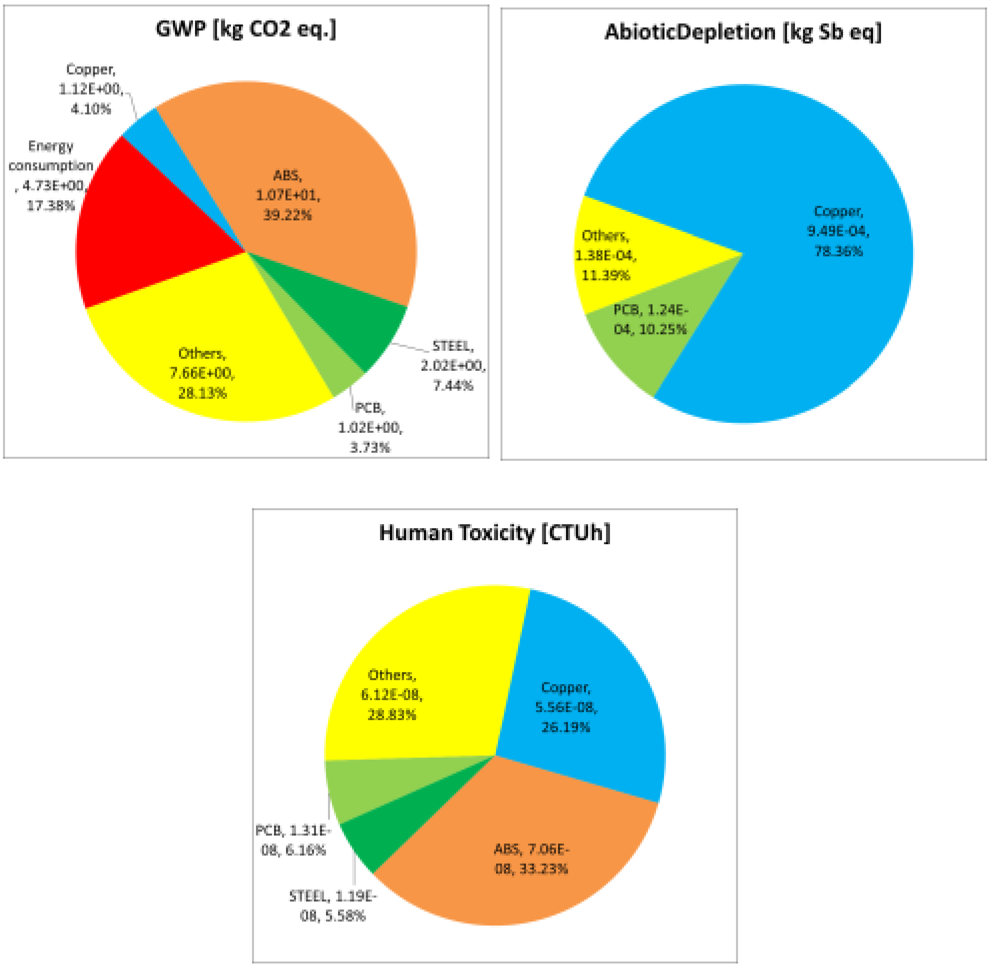
From the performed contribution analysis, it emerged that the energy consumption for the manufacturing phase is negligible (3.5% of the GWP of the VC, negligible for both HTc and ADP).

Focusing on materials contribution to the GWP, a relevant contribution (more than 35%, i.e. 10.68 kg $CO_{2eq}$) of the manufacturing impact should be associated to the ABS used for the hose, the hose reel and the casing of the canister. Note that ABS is also the most relevant material of the VC in terms of mass.

For the ADP, the most relevant components for the ADP are the motor and the printed circuit board. In particular, the greatest impact is due to the content of copper (in the rotor, stator, and in the cables), even if the content of copper is less than 10% of the motor mass). Indeed, the PCB mass contribution is lower than 0.5%, and that the major contribution of its impact is to be associated to the printing wiring board (5.45E-05 kg Sb eq.) and transistor (2.84E-05 k g Sb eq.).

Copper is also the major contributor to the HTc impact category, but also ABS has a significant share, i.e. more than 30%.

Figure 13: Contribution of the canister's materials to the overall impact of the canister vacuum cleaner

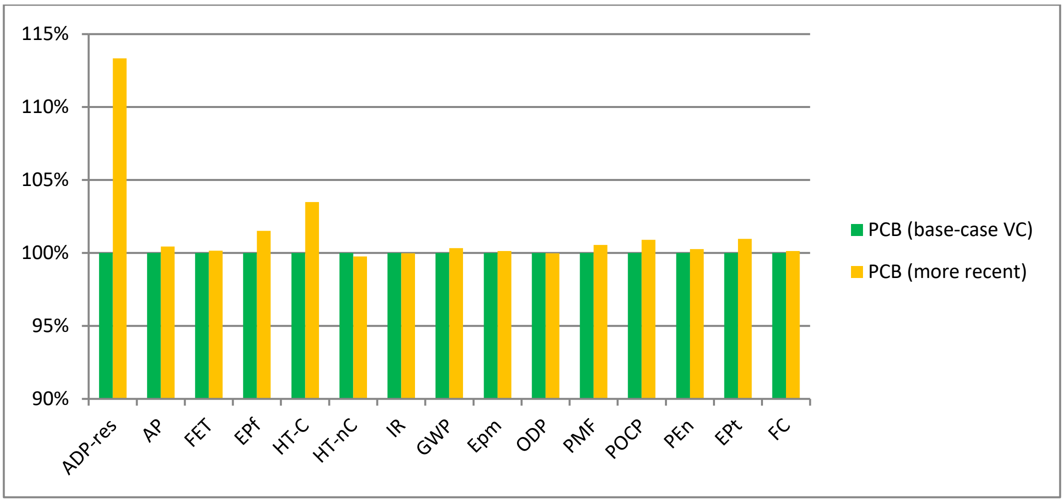



In order to take into account the potential technological development, the environmental burdens associated to the production of a new and more technologically advanced PCB used in product (B) have been considered. Due to limited data available, the sensitivity analysis was realized assuming to vary the impact of the PCB, according to a parameter γ:

$$I_{PCB'} = \gamma \cdot I_{PCB} \quad , \qquad 0 \leq \gamma$$

The range of γ variation is substantially related to the amount of many components (Bobba et al., 2015). Substituting the new PCB into the LCA model, the higher variation of the overall impact is to be associated to the ADP (more than 10%), while almost all the impact categories don't vary their environmental impact for more than 1% (Figure 14). Therefore, the γ parameter has been varied between 95% and 140% (no lower value than 100% are considered). LCIA results prove that the substitution of the PCB with a more complex one could cause variation for the life cycle impacts that are generally lower than 5%. Higher differences can be found for ADP and HTc but only in correspondence to high γ-values.

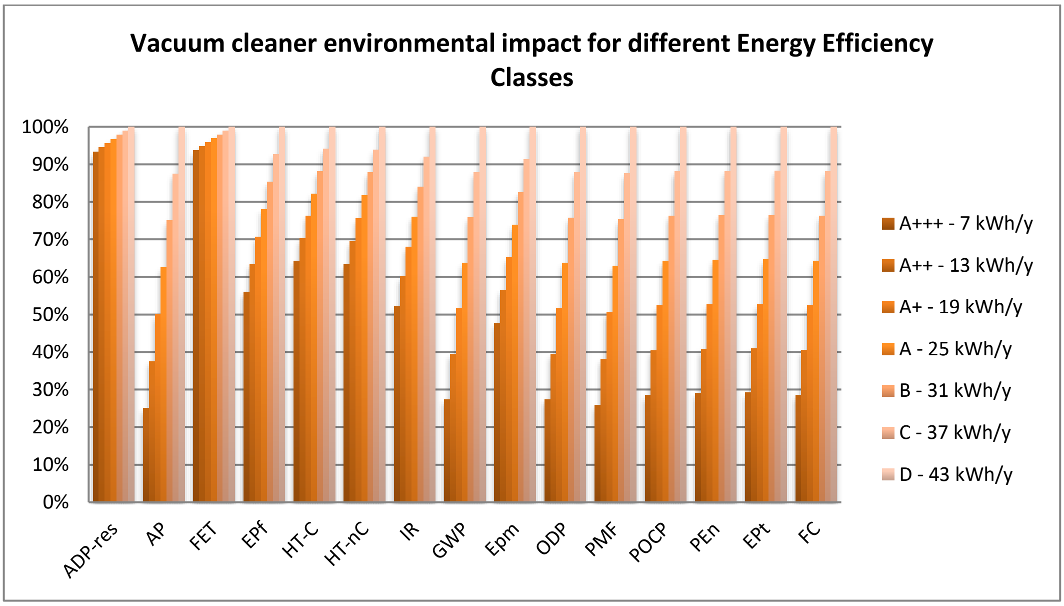
Figure 14: Comparison between the LCIAs of VC with two different PCBs.




Since the VCs is a use-phase dominant product, the energy efficiency standards in force from September, 1st 2017, were used to perform a sensitivity analysis on energy consumption, so the energy consumption of the VC is made ranging between 7 [kWh/y] and 43 [kWh/y]. LCIA results show that the environmental burdens of the electricity usage decrease with the increasing of the energy efficiency of the ErPs and the variation is particularly significant for impact categories dominated by energy (Figure 15).

A more detailed contribution analysis depicted that for ADP, the contributions of the different phases to the VC life-cycle impact do not change whichever energetic class is considered, while it is more important for both GWP and HTc.

Figure 15: Sensitivity analysis of the energy consumption (related to VC of different energy Efficiency Classes)



## 2.2.3 Environmental assessment of durability of VC

Based on the LCA illustrated in section 2.2.2, the durability assessment of the VC was performed for all the assessed impact categories. The main assumptions for the analysis and the LCA results are summarized in Table 8 and Table 9. Consistent with the hypotheses of technological development and the results of the LCIA and the sensitivity analysis (section 2.2.2.4), the adopted $\gamma$ for calculating the durability index is 105% for all the impact categories except for ADP, for which $\gamma$ = 110%.

In this section, results are discussed for three representative impact categories, i.e. GWP, HTc and ADP, are reported in this section (see (Bobba et al., 2015) for the all impact categories). These three impact categories are selected based on the obtained LCIA results:

- GWP, as dominated by the energy consumption;
- ADP , mineral resource, as dominated by the manufacturing phase;
- HTc, equally influenced by both manufacturing and use phase.

To take into account the impacts of repair operations, due to the lack of data, two different scenarios are considered: a "low-repair scenario" (LRS) and a "high-repair scenario" (HRS). It is assumed that the LRS minor operations occur and such operations do not affect the overall environmental impact of the product. In the HRS the substitution of some components is assumed. According to the LCIA results, the components that most importantly affect the life-cycle impacts, if substituted, are the PCB and the hose. Thus, they are used as reference to establish the $R_A$ value for the HRS:

- from 0% to 1% for the impact categories: Ecotoxicity for aquatic fresh water, Ionising Radiation, Particulate Matter, Ozone Depletion and Total Freshwater Consumption[35];
- from 1% to 3% for the impact categories: Acidification, Freshwater, Marine and Terrestrial Eutrophication, Human Toxicity (non-cancer effects), IPCC Global Warming, Photochemical Ozone Formation and Primary energy from non-renewable resources[36];
- between 5% and 10% for the impact categories: Abiotic depletion and Human Toxicity (cancer effects)[37].

The values of the durability index for three representative impact categories are depicted in Table 10, Figure 16 and Figure 17.

The obtained results prove that the extension of the lifetime ensures environmental benefits for all the assessed impact categories. For instance, for a lifetime extension of 100 hours, the life-cycle GWP compared to the replacement of the VC with a new one 15% more efficient[38] is reduced by 1.69%.

[35] HRS is assumed to be 1% of the manufacturing impact
[36] HRS is assumed to be 3% of the manufacturing impact
[37] HRS is assumed to be 5% of the manufacturing impact
[38] The value of 15% energetically more efficient is corresponding to about one energy efficient class higher

Table 8: Summary of the assumptions for the calculation of the durability index

| Product "A" | | | |
|---|---|---|---|
| Average operating time | $T_A$ | [hours] | 500 |
| Yearly energy consumption until 500 hours | | [kWh/y] | 25 |
| Extension of the lifetime | X | [hours] | 0 → 300 |
| **Product "B"** | | | |
| Variation of the manufacturing impact of product "B" compared to "A" | γ | [%] | γ = 105% (γ = 110% for ADP) |
| Variation of the energy consumption impact of product "B" compared to "A" | δ | [%] | 70%< δ < 100% |

Table 9: Summary of the life cycle impacts of canister vacuum cleaner

| | **IPCC global warming, excl biogenic carbon** | | **Human toxicity canc. effects, USEtox (recommended)** | | **Abiotic Depletion, mineral resources** | |
|---|---|---|---|---|---|---|
| | **[kg CO2-Equiv.]** | | **[CTUh]** | | **[kg Sb-Equiv.]** | |
| $P_{A,n}$ | 2.72E+01 | | 2.11E-07 | | 1.21E-03 | |
| $E_n$ | 1.59E+00 | | 4.99E-09 | | 9.16E-07 | |
| $U_{A,n}$ | 2.37E-01 | | 20.3E-10 | | 1.22E-07 | |
| $R_{A,n}$ | LRS | 0.0E+00 | LRS | 0.0E+00 | LRS | 0.00E+00 |
| | HRS | 8.17E-01 | HRS | 1.06E-08 | HRS | 6.06E-05 |

Note that a negative Durability Index (for low values of δ-parameter) means that, benefits related to the lifetime extension of the VC strictly depend on the energy efficiency of the new product (B). This is particularly evident for those impact categories dominated by energy consumption). Results proved that the extension of the lifetime of VCs generally produces relevant environmental benefits from a life-cycle point of view, even if the replacing product is more energy efficient. Based on this analysis, it is observed that

- The higher is the lifetime extension the higher is the environmental benefit in terms of life-cycle GWP: a lifetime extension of 250 h can reduce the life-cycle GWP of 4.23% compared to the replacement of the old product with a new one 15% more efficient, while a lifetime extension of 100 h, under same hypothesis, can reduce the life-cycle GWP of 1.69%
- Concerning the impact categories dominated by energy consumption, the Base-case Scenario (i.e. the substitution of the VC with a new more energy efficient product) discloses some environmental benefits for low values of the δ parameter (Acidification, GWP, Ozone Depletion, Particular Matter, Photochemical Ozone Formation and Total Freshwater consumption). For instance, in terms of life-cycle GWP, the Durability Index ($D_n$) is negative when the new product is 26% more efficient than the replaced one (that means the replacing VC is almost two energy efficiency classes higher than the replaced one).
- The environmental benefits are more relevant for im*pact categories dominated by the production phase (e.g. HTc and ADP).* The extension of the lifetime of the VC of 100 h can reduce the life-cycle HTc of more than 12.35% whatever the energy efficiency increase of the replacing product. Similarly, the 100 h lifetime extension can reduce the life-cycle ADP of more than 20.66.

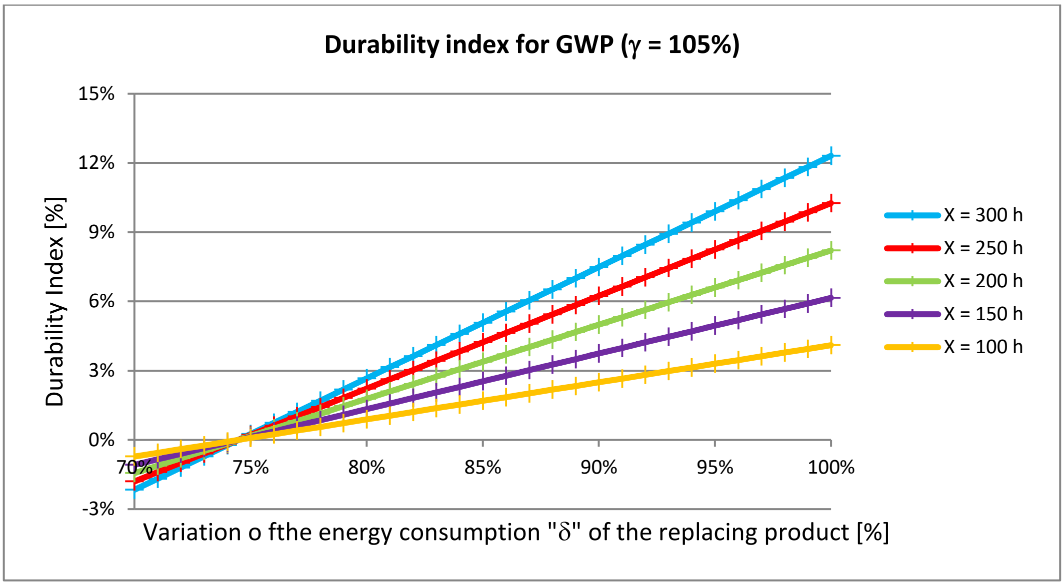
- The accounting of additional environmental impacts due to repairing (HRS scenarios) implies lower environmental benefits, especially for low values of the lifetime extension. This difference is negligible for the use phase dominant categories, while is more relevant for categories dominated by manufacturing phase.

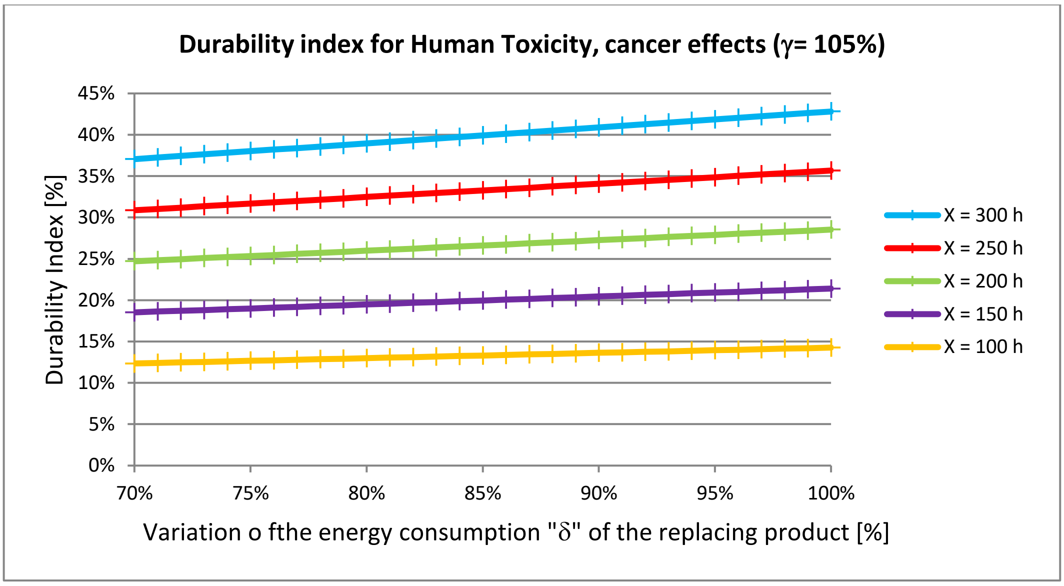
Table 10: Example of Durability index ($D_n$) results (lifetime extension of 100 hours)

| δ [%] | ADP-res | AP | FET | EPf | HT-C | HT-nC | IR | GWP | Epm | ODP | PMF | POCP | PEn | EPt | FC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 100% | 20,95% | 3,37% | 40,09% | 22,10% | 14,27% | 28,83% | 11,74% | 4,10% | 19,57% | 8,66% | 7,44% | 8,98% | 4,73% | 9,39% | 7,58% |
| 99% | 20,94% | 3,20% | 40,07% | 21,91% | 14,20% | 28,71% | 11,65% | 3,94% | 19,35% | 8,34% | 7,11% | 8,66% | 4,57% | 9,08% | 7,25% |
| 98% | 20,93% | 3,03% | 40,05% | 21,72% | 14,14% | 28,58% | 11,56% | 3,78% | 19,14% | 8,03% | 6,78% | 8,35% | 4,42% | 8,77% | 6,93% |
| 97% | 20,92% | 2,87% | 40,03% | 21,53% | 14,08% | 28,46% | 11,48% | 3,62% | 18,93% | 7,71% | 6,45% | 8,03% | 4,26% | 8,46% | 6,60% |
| 96% | 20,91% | 2,70% | 40,02% | 21,34% | 14,01% | 28,33% | 11,39% | 3,46% | 18,71% | 7,39% | 6,13% | 7,72% | 4,11% | 8,15% | 6,27% |
| 95% | 20,90% | 2,53% | 40,00% | 21,15% | 13,95% | 28,21% | 11,30% | 3,30% | 18,50% | 7,07% | 5,80% | 7,41% | 3,96% | 7,84% | 5,94% |
| 94% | 20,89% | 2,36% | 39,98% | 20,97% | 13,89% | 28,08% | 11,21% | 3,14% | 18,29% | 6,76% | 5,47% | 7,09% | 3,80% | 7,53% | 5,62% |
| 93% | 20,88% | 2,19% | 39,96% | 20,78% | 13,82% | 27,95% | 11,13% | 2,98% | 18,08% | 6,44% | 5,14% | 6,78% | 3,65% | 7,22% | 5,29% |
| 92% | 20,87% | 2,03% | 39,94% | 20,59% | 13,76% | 27,83% | 11,04% | 2,82% | 17,86% | 6,12% | 4,81% | 6,46% | 3,49% | 6,90% | 4,96% |
| 91% | 20,86% | 1,86% | 39,93% | 20,40% | 13,69% | 27,70% | 10,95% | 2,66% | 17,65% | 5,81% | 4,48% | 6,15% | 3,34% | 6,59% | 4,63% |
| 90% | 20,85% | 1,69% | 39,91% | 20,21% | 13,63% | 27,58% | 10,86% | 2,50% | 17,44% | 5,49% | 4,15% | 5,83% | 3,18% | 6,28% | 4,31% |
| 89% | 20,84% | 1,52% | 39,89% | 20,02% | 13,57% | 27,45% | 10,78% | 2,33% | 17,22% | 5,17% | 3,82% | 5,52% | 3,03% | 5,97% | 3,98% |
| 88% | 20,83% | 1,36% | 39,87% | 19,83% | 13,50% | 27,33% | 10,69% | 2,17% | 17,01% | 4,85% | 3,49% | 5,21% | 2,87% | 5,66% | 3,65% |
| 87% | 20,82% | 1,19% | 39,85% | 19,64% | 13,44% | 27,20% | 10,60% | 2,01% | 16,80% | 4,54% | 3,17% | 4,89% | 2,72% | 5,35% | 3,32% |
| 86% | 20,81% | 1,02% | 39,83% | 19,45% | 13,37% | 27,08% | 10,51% | 1,85% | 16,58% | 4,22% | 2,84% | 4,58% | 2,56% | 5,04% | 3,00% |
| 85% | 20,80% | 0,85% | 39,82% | 19,26% | 13,31% | 26,95% | 10,43% | 1,69% | 16,37% | 3,90% | 2,51% | 4,26% | 2,41% | 4,73% | 2,67% |
| 84% | 20,79% | 0,68% | 39,80% | 19,07% | 13,25% | 26,83% | 10,34% | 1,53% | 16,16% | 3,58% | 2,18% | 3,95% | 2,25% | 4,42% | 2,34% |
| 83% | 20,78% | 0,52% | 39,78% | 18,88% | 13,18% | 26,70% | 10,25% | 1,37% | 15,94% | 3,27% | 1,85% | 3,63% | 2,10% | 4,11% | 2,01% |
| 82% | 20,77% | 0,35% | 39,76% | 18,69% | 13,12% | 26,58% | 10,16% | 1,21% | 15,73% | 2,95% | 1,52% | 3,32% | 1,94% | 3,80% | 1,69% |
| 81% | 20,77% | 0,18% | 39,74% | 18,50% | 13,05% | 26,45% | 10,08% | 1,05% | 15,52% | 2,63% | 1,19% | 3,01% | 1,79% | 3,49% | 1,36% |
| 80% | 20,76% | 0,01% | 39,73% | 18,32% | 12,99% | 26,33% | 9,99% | 0,89% | 15,30% | 2,32% | 0,86% | 2,69% | 1,63% | 3,18% | 1,03% |
| 79% | 20,75% | -0,15% | 39,71% | 18,13% | 12,93% | 26,20% | 9,90% | 0,73% | 15,09% | 2,00% | 0,53% | 2,38% | 1,48% | 2,87% | 0,70% |
| 78% | 20,74% | -0,32% | 39,69% | 17,94% | 12,86% | 26,07% | 9,81% | 0,57% | 14,88% | 1,68% | 0,21% | 2,06% | 1,32% | 2,56% | 0,38% |

| | | ADP-res | AP | FET | EPf | HT-C | HT-nC | IR | GWP | Epm | ODP | PMF | POCP | PEn | EPt | FC |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | 77% | 20,73% | -0,49% | 39,67% | 17,75% | 12,80% | 25,95% | 9,73% | 0,41% | 14,66% | 1,36% | -0,12% | 1,75% | 1,17% | 2,25% | 0,05% |
| | 76% | 20,72% | -0,66% | 39,65% | 17,56% | 12,73% | 25,82% | 9,64% | 0,24% | 14,45% | 1,05% | -0,45% | 1,43% | 1,01% | 1,94% | -0,28% |
| | 75% | 20,71% | -0,83% | 39,63% | 17,37% | 12,67% | 25,70% | 9,55% | 0,08% | 14,24% | 0,73% | -0,78% | 1,12% | 0,86% | 1,63% | -0,61% |
| | 74% | 20,70% | -0,99% | 39,62% | 17,18% | 12,61% | 25,57% | 9,46% | -0,08% | 14,03% | 0,41% | -1,11% | 0,81% | 0,70% | 1,32% | -0,93% |
| | 73% | 20,69% | -1,16% | 39,60% | 16,99% | 12,54% | 25,45% | 9,38% | -0,24% | 13,81% | 0,10% | -1,44% | 0,49% | 0,55% | 1,01% | -1,26% |
| | 72% | 20,68% | -1,33% | 39,58% | 16,80% | 12,48% | 25,32% | 9,29% | -0,40% | 13,60% | -0,22% | -1,77% | 0,18% | 0,39% | 0,70% | -1,59% |
| | 71% | 20,67% | -1,50% | 39,56% | 16,61% | 12,41% | 25,20% | 9,20% | -0,56% | 13,39% | -0,54% | -2,10% | -0,14% | 0,24% | 0,39% | -1,92% |
| | 70% | 20,66% | -1,66% | 39,54% | 16,42% | 12,35% | 25,07% | 9,11% | -0,72% | 13,17% | -0,86% | -2,43% | -0,45% | 0,08% | 0,08% | -2,24% |

Figure 16: Durability index for the canister vacuum cleaner (LRS scenario)



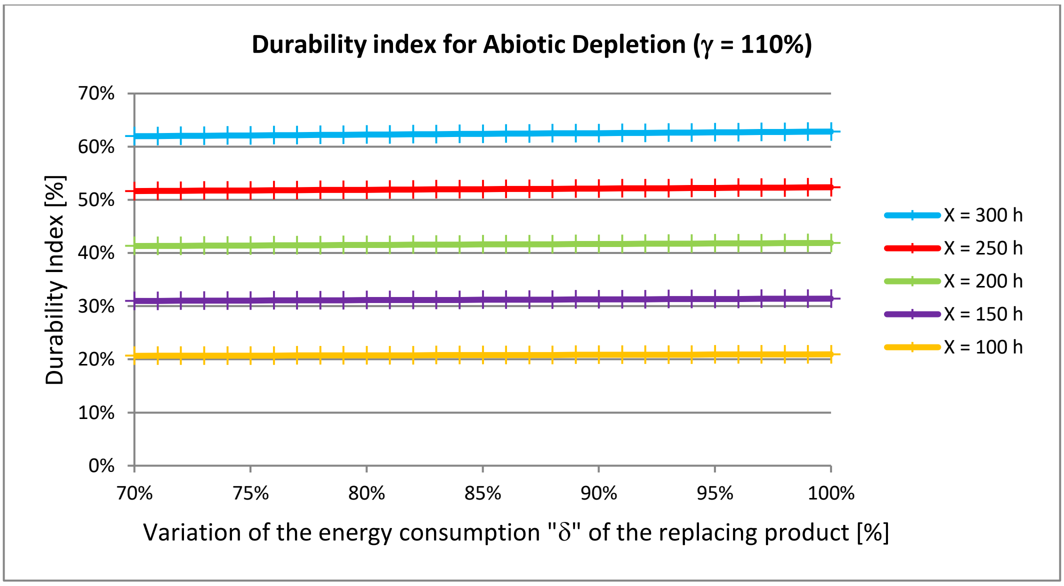
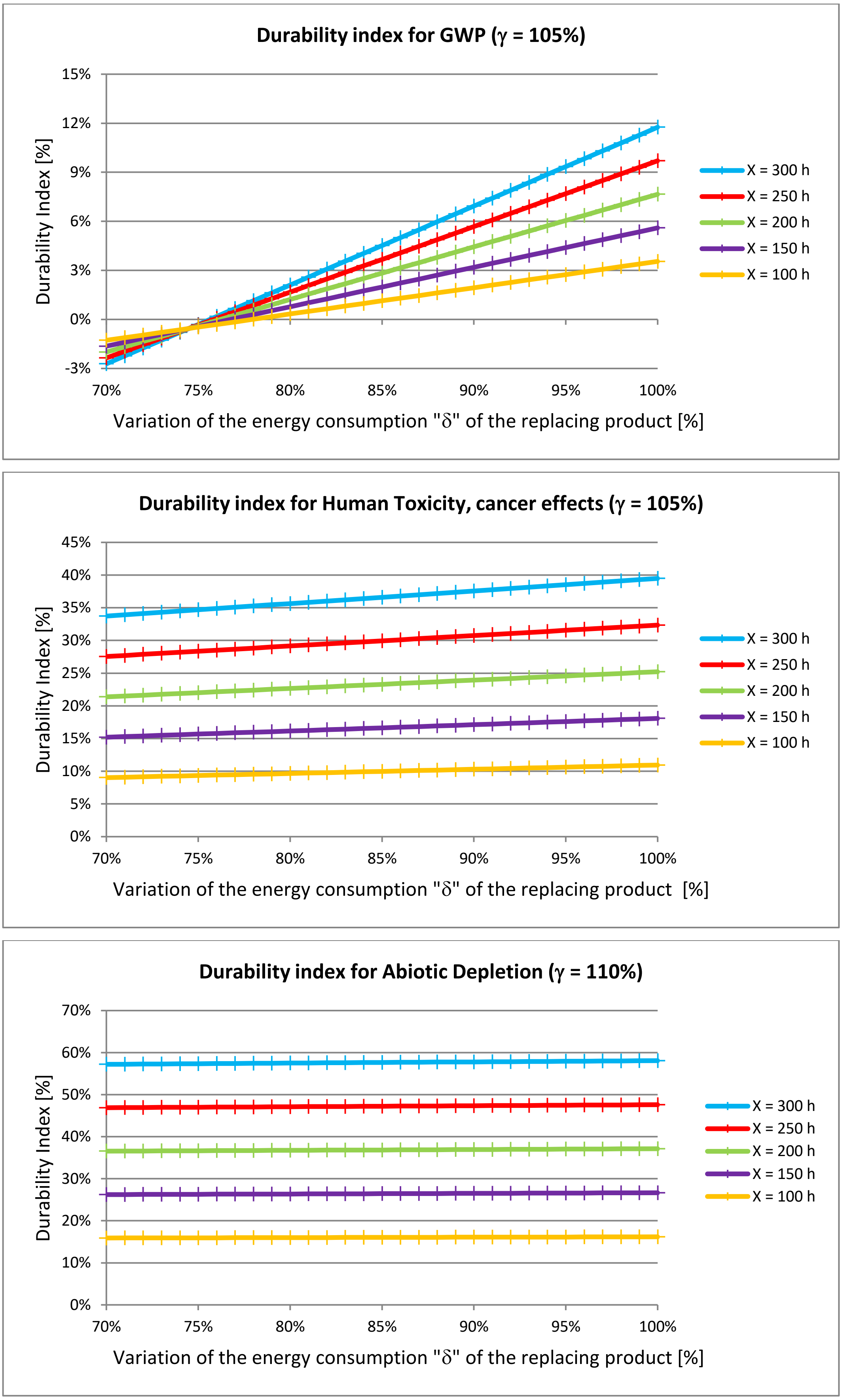



Figure 17: Durability index for the canister vacuum cleaner (HRS scenario)



### 2.2.3.1 Further analyses

Results are then compared with the durability assessment of VC adopting the Ecoreport tool[39], a tool developed within the "Methodology for Ecodesign of Energy-related Products" (BIO by Deloitte, 2013). For the comparison, the same input data used for the LCA of the case-study VC were used.

Some limitations are related to:

- absence of materials available in the tool. Therefore, some extra materials have been inserted *ad hoc*, i.e. glass fibre (BMC-GF) and polyoxymethylene (POM) of the canister case and motor parts.
- Concerning the maintenance, the Ecoreport tool models the use of dust-bags as auxiliary materials, but not the filters; thus, these have been inserted among the input materials list as part of the manufacturing.
- Concerning EoL, materials are modelled according to macro-groups[40] and not by product component. Therefore, it is not possible to differentiate the EoL scenario for each material. For the present analysis, it was considered an average "recycling rate" for the materials belonging to a specific group[41].

Note that a more in depth analysis of the impact categories concluded that not all the impact categories in the Ecoreport tool (Kemna, 2011a, 2011b; Kenma et al., 2005) can be compared to the impact categories adopted in the developed assessment. Table 11 summarizes the impact assessment results with both the Ecoreport tool and the LCA (section 2.2.2). The detailed comparison between the impact categories is provided in (Bobba et al., 2015).

Table 11: Difference between the Ecoreport Tool life-cycle impacts and those calculated with the impact categories. The red cells identify the impact categories with the highest "Variation", while the green cells the categories with the lowest "Variation".

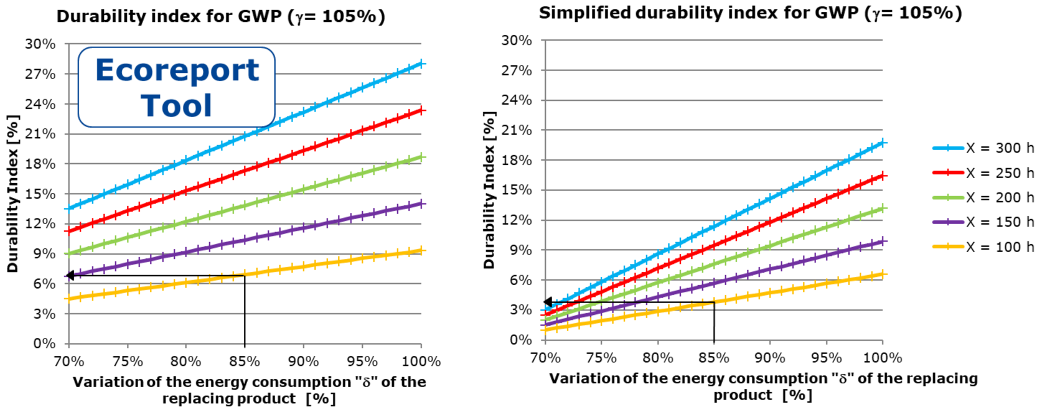
| Impact category | Ecoreport tool | LCIA | Variation[42] |
|---|---|---|---|
| Total Energy (GER) | 3,112 [MJ] | 2,923 [MJ] | 6.07% |
| Water (process) | 1,219 [l] | 451,336 [l] | -36,909.92% |
| Greenhouse Gases in GWP100 | 137 [kg $CO_2$ eq.] | 149 [kg $CO_2$ eq.] | -8.74% |
| Acidification, emissions | 0.64 [kg $SO_2$ eq.] | 0.71 [kg $SO_2$ eq.] | -11.3% |
| Volatile Organic Compounds (VOC) | 0.26 [kg $_{VOC}$] | 0.32 [kg NMVOC] | -24.46% |
| | | 0.03 [kg] | 87.18% |
| Particulate Matter (PM, dust) | 0.20 [kg $_{PM, dust}$] | 0.17 [kg PM10 $_{eq}$] | -13.48% |
| Eutrophication | 4.33E-03 [kg $_{PO4}$] | 4.35E-02 [kg $_{PO4}$.] | -904.34% |

[39] The Ecoreport tool is available at http://ec.europa.eu/enterprise/policies/sustainable-business/ecodesign/methodology/index_en.htm (accessed March 2015)
[40] For example, plastics are grouped into Bulk plastics and Tec Plastic.
[41] For example, for bulk plastics, it was calculated a recycling rate as average of the amount of recycled LD-PE, HD-PE, LLD-PE, PP, PS, EPS, HI-PS, PVC, SAN, PET, ABS
[42] Calculated as: (Ecoreport – LCIA)/ Ecoreport [%]

The impact assessment results obtained with the Ecoreport tool are then used to estimate the Durability index according to the illustrated method. The comparison focused of the GWP impact category. Results trends (Figure 18) are similar to those obtained from the performed LCIA even though the index calculated using the Ecoreport tool as input data has higher values of the index compared to those obtained through the performed LCA.

Overall, the Durability index can be easily implemented/calculated with the values derived from the Ecoreport tool.

Figure 18: comparison of the Durability index based on the Ecoreport tool (left) and on the performed LCA (right)




Finally, to take into account the uncertainty of some of the input data, a **sensitivity analysis** was performed to estimate the effect of some parameters on the overall results. In particular, it is observed that, focusing on *the parameters "γ" and "α"*:

- these variations of the parameters "γ" and "α" do not largely affect the "$D_n$" index for the GWP impact, whereas some variations are observed for the ADP and HTc impacts[43]
- the variation of "$D_n$" is higher for higher extension of the lifetime (X)
- the additional impacts due to the production of more durable products ("α") is more relevant for high value of lifetime extension. However, the variation of "α" does not largely affect the "$D_n$" when referring to the ADP impact category.

## 2.2.4 Economic assessment of durability of VCs

To assess the potential economic benefits related to extend the lifetime of VC through their repair, the necessary input data were collected from the available literature, also focusing on Eurostat statistics and available market data. The main assumptions on VC characteristics and market prices are summarized in Table 12. Note that in defining parameters, it is assumed that in case of high cost of repair, consumers would buy a new VC instead of repairing the old one.

[43] Considering a lifetime extension of 250 h, the GWP varies less than 0,5% when varying "γ" by ±2%. The ADP varies up to 10% when "γ" varied by ±20%.

Table 12: Summary of the assumptions for the calculation of the $\Delta C_{TOT}$

| Parameter | Value | Note |
|---|---|---|
| Average operating time ($T_A$) | 10 [years]<br>500 [hours] | |
| Yearly energy consumption until 500 hours | 25 [kWh/y] | |
| Extension of the lifetime (X) | 0 →300 [hours] | |
| Variation of the manufacturing impact of product (B) compared to (A) ($\gamma$) | $\gamma$ = 105%<br>($\gamma$ = 110% for ADP) | For the sensitivity analysis $\gamma$ is assumed to vary in the range 103% ≤ $\gamma$ ≤ 107%<br>(90% ≤ $\gamma$ ≤ 130% for ADP) |
| Variation of the energy consumption impact of product (B) compared to (A) ($\delta$) | 70% < $\delta$ < 100% | |
| Variation of the additional impacts of durable product ($\alpha$) | $\alpha$ = 1% for GWP<br>$\alpha$ = 4% for ADP<br>$\alpha$ = 7% for HTc | For the sensitivity analysis $\alpha$ is assumed to vary in the range: 0% ≤ $\alpha$ ≤ 2% for GWP; 3% ≤ $\alpha$ ≤ 5% for ADP; 6% ≤ $\alpha$ ≤ 8% for HTc, |
| Price of product (A) ($C_A$) | 150 [€] | Information about purchasing price of VCs were collected by the following sources: (AEA, 2009; Wollerton, 2013).<br>For the sensitivity analysis $C_A$ is assumed to vary in the range 100€ ≤ $C_A$ ≤ 200€ |
| Price of product (B) ($C_B$) | (1+$\beta$) • $C_A$<br>$\beta$ = 20% | For the sensitivity analysis $\beta$ is assumed to vary in the range -15% ≤ $\beta$ ≤ 25% |
| Price of more durable product (A') ($C_{A'}$) | 170 [€] | For the sensitivity analysis $C_A$ is assumed to vary in the range 150€ ≤ $C_A$ ≤ 200€ |
| Price of electricity ($El_{t_1}$) | 0.205 [€/kWh] | Information about purchasing price of VCs were collected by the following sources: (EUROSTAT, 2015) |
| Growth rate of electricity price | 4% | Information about purchasing price of VCs were collected by the following sources: (EC, 2014b)<br>For the sensitivity analysis the growth rate of electricity price is assumed to vary in the range 1% ÷ 7% |
| Discount rate (i) | 3% | Information about purchasing price of VCs were collected by the following sources: (Iraldo and Facheris, 2015).<br>For the sensitivity analysis "i" is assumed to vary in the range 1% ≤ i ≤ 5% |
| Repair costs (R) | 20% • $C_A$ | The repair expenditures will occur after 11 years lifetime of product A.<br>For the sensitivity analysis R is assumed to vary in the range 0% • $C_A$ ≤ R ≤ 40% • $C_A$ |
| Auxiliaries costs (AU) | 1.75 [€/dustbag] | For the sensitivity analysis A is assumed to vary in the range 1.5€ ≤ AU ≤ 2€ |
| Maintenance costs (M) | 2 [€/set of filters] | For the sensitivity analysis A is assumed to vary in the range 2€ ≤ M ≤ 14€ |

The life-cycle costs results are reported in Figure 19. Longer lasting VC represents the most viable option: despite the higher energy efficiency of the new product. After 600 hours (i.e. 1 year of lifetime extension), the repair cost occurs: in fact, the total cost increase of about 22 €, that represents less than 6% of life-cycle costs. In any case, the total cost of the Durability scenario is always lower than the base-case ones, consistent with the previous discussion.

Note that, the electricity price has almost the same contribution (between 16% and 18% of the life-cycle costs) irrespective of the lifetime extension options or the price growth of the new product

Figure 19: Life Cycle Costs (LCC) of the Base-case (first column) and the Durability scenario (second column)

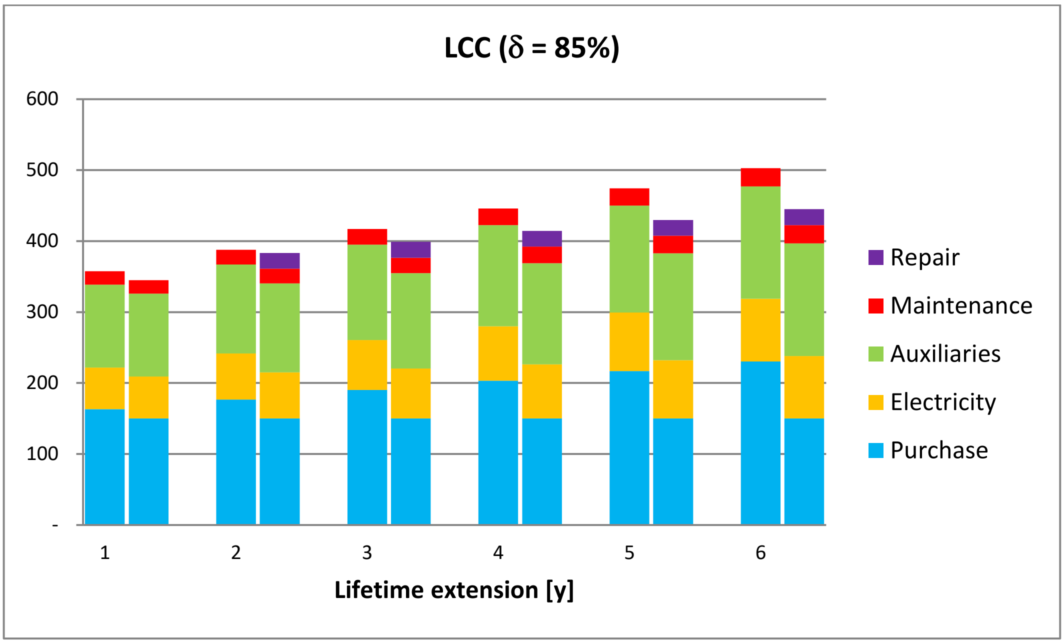



Figure 20 shows the difference between the Base-case scenario and the Durability scenario, with the lifetime extended up to 11 years (550 hours). Extending the lifetime of the base-case VC to 550 h, the gain is of about 11-13 €, depending on δ. Note that in this case R=0.

When repair costs are included in the analysis for longer than 1 year lifetime extension, results slightly change as illustrated in Figure 21. It is observed that:

- The repair costs to prolong the lifetime beyond 500 hours of operating time make the Durability scenario less viable from the economic point of view.
- Despite the repair costs, the Durability scenario has economic benefits when the lifetime is increased by more than 1 year: the higher is the extension of the lifetime the higher are the benefits.

Due to the uncertainty of data, a **sensitivity analysis** was made varying the following parameters (using a fix value of δ = 85%):

- Purchase price of the base-case product ($C_A$);
- Purchase price of the replacing product ($C_B$);
- Repair costs (R);
- Auxiliaries costs (AU);
- Maintenance costs (M);
- Discount rate (i);
- Electricity price growth rate.

Results proved that the variation of the discount rate (i), the growth rate of electricity and the auxiliary materials (A) do not significantly affect the final results. On the other hand, the variation of the repair costs (R), the maintenance costs (M) and of the assumptions about the purchase prices of the products (C) are more relevant. In fact, R can make the Durability scenario not convenient from the economic point of view. For example, assuming the replacing product (B) 15% more efficient than the product (A), and assuming the repair costs (R) equal to 30% of the purchasing price of the product (A), the Durability scenario is convenient from an economical perspective only if the extension of the lifetime is higher than 130 h (i.e. 2.6 years).

Figure 20: $\Delta C_{TOT}$ for the canister vacuum cleaners with considering a lifetime extension of 1 year (i.e. a lifetime of 500 h + 50 h)

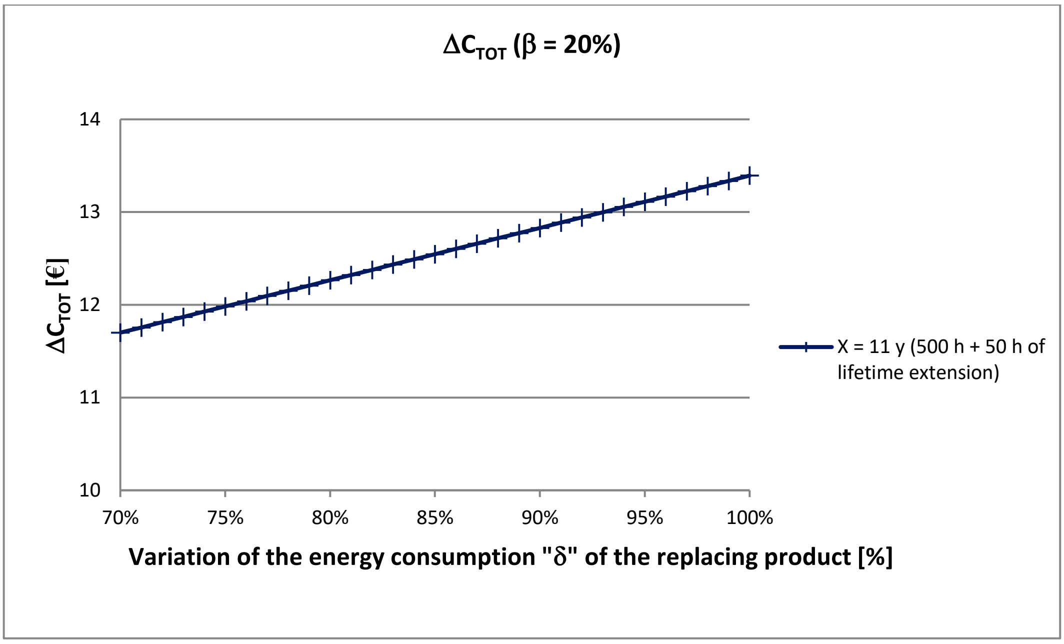
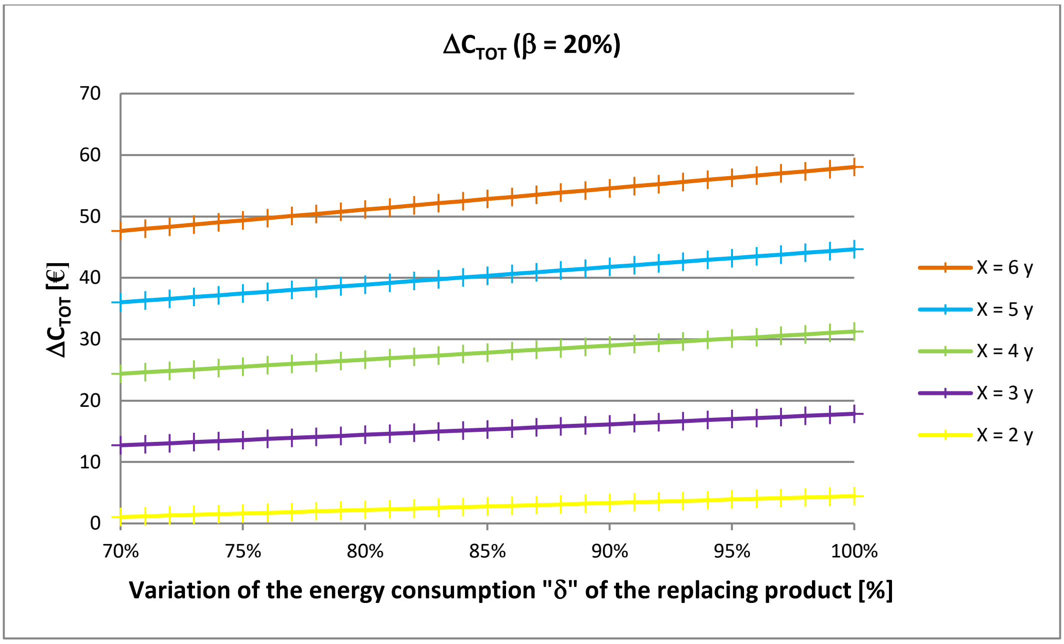



Figure 21: $\Delta C_{TOT}$ for the canister vacuum cleaners (on varying X-values)



## 2.3 Lesson learnt and follow up

Despite the LCA is recognised as a suitable tool to assess the environmental performances of products along their life-cycle, to better understand the environmental performance of durability, a broader analysis is needed and LCA is used as the background for the analysis. Then, the assessment of potential environmental benefits/drawbacks related to the extension of lifetime of products was performed based on the REAPro method (Ardente and Mathieux, 2014b). However, simplifications were used and a more general method was developed in order to capture some aspects not included by those authors, e.g. the technological development expected in the next years. To also capture economic aspects, which are recognised as significant in the potential adoption of longer lasting products, environmental assessment of durability was complemented by an economic assessment of durability. Therefore, a method to assess the economic performances of extending the lifetime of products was developed, following the same approach used for the environmental assessment method. These two assessments are part of the "Pro-EnDurAncE" (Environmental and Economic Assessment of Durability of Products) method, which could be potentially applied to various product.
The Pro-EnDurAncE was then applied to a case-study product selected among one of the most relevant sectors in Europe in terms of environmental impacts as identified by the EIPRO study: "housing" (Tukker et al., 2006). Among products belonging to this sector, it is applied to the "VC" product group. Therefore, an LCA of a specific VC was performed and then used as input for the durability assessment. Due to the lack of data, primary data were collected through the dismantling of a canister VC and complemented by literature and market data, when necessary.

Results proved that the environmental benefits can occur when extending the lifetime of VCs, especially concerning impact categories mainly affected by energy consumption. However, the manufacturing phase is not negligible for all the impact categories, especially if changes related to the technological development (e.g. more complex products, different materials, different manufacturing processes) will occur in the next future. Compared to the environmental assessment, in the economic assessment, the relevance of some components emerged. This is the case for instance of auxiliaries and maintenance. Despite their contribution to the life-cycle environmental impacts is not so relevant, their contribution to the life-cycle costs is quite important. Concluding on the VC product group, promoting the design of more durable VC can have benefits from an environmental perspective. Moreover, cost considerations can be used to identify critical aspects for users and manufactures and potentially incentives. The uncertainty of specific parameters would require a sensitivity analysis based on data provided by stakeholders of the value chain, e.g. manufacturers, repairer, sellers.

The method is general enough to be applied to different products, to perform different analysis and assess various scenarios according to the available input data. Moreover, the combination of different aspects assessed in the same method allowed having a more complete overview of the impacts of extending the lifetime of products from different perspectives. The application to the specific case-study proved the *relevance of having a multi-criteria approach for the assessment of durability.*

# Application of the modular Life Cycle Assessment to the automotive sector

The environmental assessment of products along their whole life-cycle requires detailed information on each life-cycle step. In the proposed methodological framework, LCA is adopted to provide the necessary background information of the products/services, which is particularly relevant in case of complex products that also have a relevant contribution to the environmental impact in Europe, as for vehicles. The application of the LCA in the automotive sector is not a novelty since different stakeholders already adopted the LCA since Nineties (BOX 4). The main novelty proposed in this chapter is the direct link between the LCA model (especially Life Cycle Inventory) with a database where car manufacturers store data of different models of vehicles. The link between the structured data of industrial stakeholders directly operating in the automotive sector and a flexible LCA model could offer the opportunity to assess the environmental performances of different vehicles and assure the consistency of comparison between different vehicles.

As mentioned in previous chapters, the assessment of the lifetime extension of products, especially in case of the comparison of different scenarios, should consider the existence of different products available in the market. Moreover, consistent with the debate on benefits of reuse (section 1.4), the technological development and the potential introduction of new materials/products/components are key aspects to be considered in the assessment. This also emerged from chapter 2 and the application of the Pro-EnDurAncE method to the case-study VC (section 2.3), where some parameters used in the assessment focus on the future changes in terms of specific components (e.g. new material or different quantitates), of energy (e.g. more efficient processes, different energy mix) and new technologies (e.g. new recycling processes to recover different and/or more materials). In general, the higher is the complexity of the assessed system/product, the higher is the effort in performing robust LCAs focusing on various alternatives, e.g. in case of various design options or comparison between different products (Gabrisch et al., 2019).
In this perspective, when focusing on complex products that are affected by fast technological changes, the modular set-up of LCA models offers the possibility to assess different types of products, speeding-up the comparison of LCA outcomes without performing *ad hoc* LCAs. Also, when studying products for which there exist databases with structured information, this is an incentive to develop modular LCAs, e.g. to support design of products (Brondi and Carpanzano, 2011) or decision-making (Haupt et al., 2018).

Despite the complexity of performing robust and transparent LCAs in the automotive sector, the potential of a Life Cycle Thinking approach is pointed out by several studies to evaluate the environmental impacts and the potential improvement of the technological development (e.g. implementation of regulations, development of product, introduction of new technologies, fulfilment of recycling processes) (Bauer et al., 2015; Delogu and Berzi, 2011; Nemry et al., 2008). One of the main barriers highlighted by several LCA practitioners in the automotive sector is the lack of robust data, mostly due to confidentiality of information but also the long value chain of vehicles components, which make difficult to obtain the needed data. In fact, even if the scientific literature and the public LCA often illustrate the LCIA results, no complete inventory are available; data are often aggregated or not publicly available. Therefore, a strict collaboration with stakeholders on the value-chain of vehicles is required to have access to reliable and robust data. The relevance of assessing life-cycle impacts in the automotive sector is also emphasized by the fact that on October, 3rd, the European Parliament communicated that car manufacturers have to communicate the life-cycle impacts of vehicles from 2020[44].

Thanks to the collaboration between Politecnico di Torino and *Centro Ricerche FIAT* (CRF), primary data of different types of vehicles were provided. Concerning the manufacturing phase, the main source of information was the International Material Data System (IMDS)[45], i.e. the "*automobile industry's material data system*" where all materials that are used for the manufacturing of vehicles are stored and maintained. For the other life-cycle stages, the knowledge of CRF and their structured data collection supported the development of the LCA model. According to the above-mentioned considerations, the LCA was set-up as a modular LCA and parameters were used to allow the replicability of the study, to speed-up the performance of LCA of different models and to ease the possible addition of components and or modules due to the technological development. This is highly relevant for the automotive sector due to its fast change towards a "*clean, competitive and connected mobility*" (EC, 2017b)(BOX 4).

To this aim, the LCA modelling is made up of different modules that are combined together according to the specific characteristics of the considered products. In case some modules are not suitable for the assessed product, they can be called off. To build the modules of a specific process, the inventory should include all the possible inputs/outputs of the process. Transports before and after a specific process could mark the beginning and the end of such a process (Haupt et al., 2018).

The adoption of parameters related to the inputs/outputs of each module increases the flexibility of the LCA model, makes it updatable and usable to assess different products. According to the characteristics of the assessed product, not necessarily the full list of inputs/output has to be filled in; in case of lack of data or not suitable inputs/outputs, the parameters is simply considered null. In addition, in case of lack of data or uncertain inputs/outputs, the parameterization of the model could be easily used to run sensitivity analyses. To give an example based on the previous chapter, if bag-less VCs are assessed, the full list of inputs/outputs related to the manufacturing of the case-study VC can be update with the available data and the modules/parameters related to the "auxiliaries" will be null.

In this section, relevant aspects to perform LCA of vehicles are described. In particular, methodological aspects to be considered in performing LCA analyses are illustrated in section 3.1, while the developed modular and parametrized LCA model of 4 different vehicles is described in section 3.2. Since new vehicles are entering in the market, this LCA represents a background model that could be enlarged to perform LCA of new types of vehicles, e.g. electric vehicles (xEV). Indeed, the fast penetration rate of xEV in Europe is increasing the attention of various stakeholders of the automotive sectors to the sustainability of this option. The main differences between conventional vehicles and xEV is the electric powertrain, and especially the traction batteries is recognised as the key components to further increase the penetration of xEV in

[44] http://www.europarl.europa.eu/sides/getDoc.do?pubRef=-//EP//TEXT+TA+P8-TA-2018-0370+0+DOC+XML+V0//EN

[45] http://www.mdsystem.com/imdsnt/startpage/index.jsp

Europe. Based on this, Section 3.3 reports some relevant aspects concerning the technological development of the automotive sector, especially focusing on traction batteries that should be consider in future LCA of xEV. Finally, section 3.4 collects the main outcomes of the performed analysis and the recommendations for future work.

## 3.1 LCA in the automotive sector – methodological aspects

Even though various tools are available to assess the environmental performances of the automotive sector (BOX 5), for this purpose LCA is often used. Also, LCA is used to compare environmental performances of different vehicles along their life-cycle. However, difficulties in having primary data due to the confidentiality of information and to the great number of stakeholders involved in the process (e.g. components' manufacturers). The collection of information has difficulties at different levels. First of all, LCA is barely recognised in-house and therefore availability of public and robust data for LCA studies is lacking. Moreover, the value-chain of vehicles is very complex and different actors manufacturing components around the world are part of it. Moreover, fast changes in technologies and the confidentiality of information make very difficult to have data of new components (e.g. materials embedded in specific components, content of materials in batteries, CRMs in magnets, etc.). This results in assumptions and use of available datasets used in LCA studies. Hence, the adoption of assumptions to model some life-cycle phases and the comparison between studies are complex.

Since the early 1990s, European projects were involved several European car manufacturers and their suppliers (BOX 6). In 2012, the European Automotive Manufacturers Association (ACEA) (ACEA, 2012) drew up a document regarding its position about LCA applied to automotive sector. ACEA stressed the importance of applying the LCA in the automobile industry as an internal instrument for environmentally orient the product and processes. Equally important is the diffusion of environmental performance of them by means of publications and to joint automobile industry studies with questions of general interest.

**BOX 4: Evolution in the automotive sector**

Globally, the increasing awareness concerning the environmental and potential effects related to a non-sustainable development had significant consequences in the automotive sector. Significant technological improvements allow improving the environmental performances of vehicles all over the world.

Around one quarter of Europe's greenhouse gas (GHG) emissions derive from the automotive sector (EEA, 2016). Emissions in this sector were historically dominated by passenger vehicles, also in relation to their shorter lifetime compared to other mean of transports, e.g. aircraft, trains and ships (EEA, 2018b). Europe is moving towards a more sustainable transport system and accordingly, the adopted strategy for low-emission mobility entails "Moving towards zero-emission vehicles" and "Speeding up the deployment of low-emission alternative energy for transport" (https://ec.europa.eu/transport/themes/strategies/news/2016-07-20-decarbonisation_en, (Alonso Raposo et al., 2019).

The expected decarbonisation of Europe is driving some relevant changes in the automotive sector. One of the main driver of the change refers to the $CO_2$ and air quality objectives (EES_Full_Pack + https://ec.europa.eu/jrc/en/geco, (Skinner et al., 2010)). By 2050, the EU has the goal to reduce 60% of its GHG emissions compared to 1990 (EC, 2011b). Moreover, the EU strategy also aim at decreasing the oil import dependency, increasing the innovation and competitiveness and fostering opportunities for growth and jobs.

Together with renewable biofuels, and improved use of non-motorized transport as well as public transport, e-mobility is considered one of the key technologies for the decarbonisation of the EU. This is related to the less emissions related to EV compared to traditional ICEVs: low-emission vehicles are identified within the EU legislation vehicles with tailpipe emissions lower than 50g/km. Member States are required to implement common standard to reach the European targets.

Assuming a life-cycle perspective in assessing the environmental impacts of vehicles should also consider that impacts derive from the manufacturing of some components previously not used in ICEV (Thomas et al., 2018). In fact, in this shift towards e-mobility, relevant changes in terms of powertrains and materials used for their manufacturing have to be considered. This results in the increasing relevance of both the manufacturing and the EoL stages in terms of life-cycle environmental impacts of vehicles. In fact, some authors highlighted that, whereas the most relevant contribution to the life-cycle impact of ICEV is to be attributed to the use phase, a relevant contribution of the xEV impact concerns the manufacturing stage, especially due to the powertrains manufacturing (Rotter, 2017) (Castro et al., 2003; Hawkins et al., 2013; Messagie et al., 2014a; Nordelöf et al., 2014; Notter et al., 2010; Tukker et al., 2006)

The simply replacement of ICEV with xEV will not solve other related environmental issues. One of these issues is that significant amount of raw materials are required to manufacture new vehicles, and in comparison to ICEV, the manufacturing of xEV requires different materials that are not available in big quantities in Europe. Moreover, some of these materials can belong to the CRMs list (BOX 2).

**BOX 5: Available tools assessing the environmental impacts of vehicles**

The performed literature review highlights that not only LCA is used for assessing the environmental performances of products. In fact, specific tools have been developed by several organizations/industries in order to focus the environmental assessment on vehicle.

This is the case of DC DfE-Tool, i.e. DaimlerChrysler Design for Environment (Finkbeiner et al., 2006). In this case, LCA for both components and the entire vehicle is used as an important tool of the design for the environment. Similarly, the Life-cycle Emission Model (LEM) (Delucchi, 2003) is used for the estimation of energy use, criteria pollutant emissions and GHG emissions (http://www3.epa.gov/airquality/urbanair/). Economic Input-Output, Life Cycle Analysis (EIOLCA) tool is used in (Maclean and Lave, 2003) and (Dave, 2010) to inventory resource use, environmental discharges, and economic impacts (http://www.eiolca.net/). The GREET (Greenhouse Gases, Regulated Emissions, and Energy Use in Transportation, https://greet.es.anl.gov/, https://greet.es.anl.gov/greet/index.htm) model GREET calculates emissions ($CO_2$, $CH_4$, and $N_2O$) as well as other criteria pollutants that result from transportation life cycles, taking into account life cycles of electricity, transportation fuels, and vehicle components (Argonne National Laboratory, 2014).

Other examples are the Ecoscore methodology, the Green Book ACEEE (American Council for an Energy-Efficient Economy) the Green Vehicle Guide of EPA, the Cleaner Drive of the consortium coordinated by Energy Saving Trust, the consumer guide of ETA (environmental Transport Association). Depending on the selected methodologies, specific life-cycle steps (both for the fuel and the vehicle life-cycles) are included in the environmental assessment.

Finally, the WorldAutoSteel (http://www.worldautosteel.org/) realized two specific tool (www.autolca.com):

- Vehicle Materials Energy Model (also called UCSB GHG Automotive Materials Comparison Model): Version 4.0 developed by the Santa Barbara (UCSB) University; it is an excel model which quantify energy and GHG emissions (http://www.worldautosteel.org/life-cycle-thinking/greenhouse-gas-materials-comparison-model/);
- Design Advisor: model permitting to analyse materials used in vehicles, with particular reference on mass, costs and GHG emissions. It has been developed by the Michigan University (http://www.worldautosteel.org/projects/design-advisor/).

**BOX 6: Examples of projects on LCA in the automotive sectors**

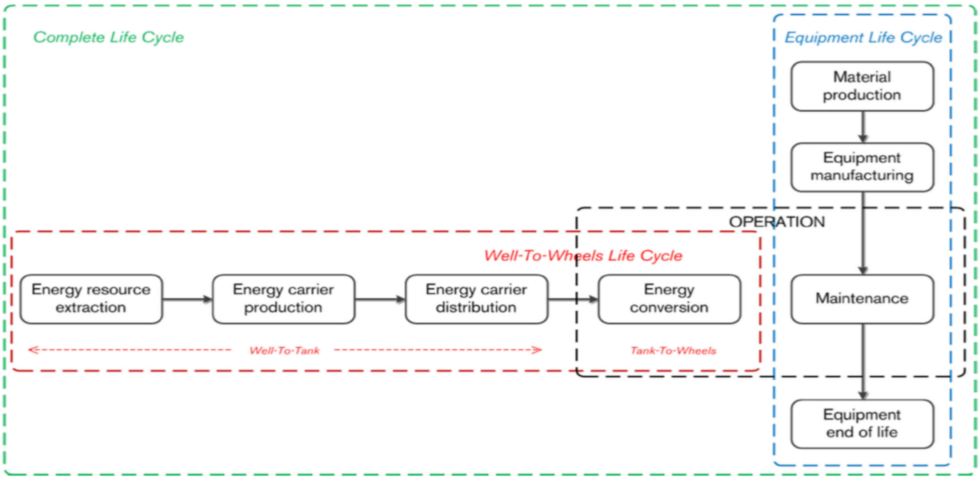
- In 1989, the University of Stuttgart (IKP) launched a program with about forty companies (nine of them car manufacturers) and their suppliers in order to develop a LCA methodology applicable to the automotive sector and a software GaBi. This project ended in 1995 (Ecobilan S.A., 1996).
- In 1992, the European Council for Automotive Research (EUCAR) launched a research program on Low Weight Vehicles. The LCA study was part of this program which was performed by a few European car manufacturers. This project ended in 1995 (Ecobilan S.A., 1996).
- In 1993, the EUCAR launched a research program on LCA, with the aim to work out a common LCA methodology for the car industry, to develop means to preserve the environment by analysing the total life cycle of existing products (e.g. energy consumption and waste reduction), to participate in environmental discussions at national and European levels. The tasks to achieve these goals were mainly two: build a reliable database and work out common guidelines to make LCA studies compatible and comparable. In fact, the project was split up in two phases. The phase 1 coordinated by PSA and ended in 1995, regarding the use phase. The phase 2 coordinated by Rover, ended in 1998, regarding the LCA guidelines (Ecobilan S.A., 1996).
- In 2006, the project “Environmental Impacts of Products” (EIPRO) (Tukker et al., 2006) was launched by European Commission with the objective to identify the products that have the greatest environmental impact throughout their life cycle, from cradle to grave. The research results identified three areas as having the greatest impact: food/drink, private transport and housing. The contribution of passenger transport to the total environmental impacts of private consumption ranges is from 15 to 35%. The greatest impact is from cars, despite major improvements in the environmental performance in recent years, especially on air emissions.
- In the year 2008, the second scientific contribution to the European Commission’s Integrated Product Policy framework was started to seek to minimize the environmental degradation caused the life cycle of products. The project is called “Environmental improvement potential of passenger cars” (IMPRO-car) (Nemry et al., 2008) and it followed a previous study coordinated by the JRC, EIPRO study. This report presents a systematic overview of the life cycle of cars, from cradle to crave. It also provides a comprehensive analysis of the technical improvement options that could be achieved in each stage of a car’s life cycle. The report has focused on the technical improvements related to the design of cars, such as the reduction of weight, improvement of the power train, reduction of rolling resistance of tires. It also analyses improvements that rely on the driver’s behaviour as speed control and eco-driving. The report examines each of the options taking into account the technical potential, the existing legislation and policy developments, and the barriers and drivers for the implementation of the different options. The study presents the consequences that the adoption of these options might have on the environment such as global warming, generation of solid waste, acidification, energy consumption.
- During 2012, the E-Mobility Life Cycle Assessment Recommendations (eLCAr) (Duce et al., 2013) project was developed to provide the guidelines for the Life Cycle Assessment (LCA) of electric vehicles and to build on the framework established by the International Reference Life Cycle Data System (ILCD). The aims were to support LCA practitioners in the European Green Cars Initiative (EGCI) and to create a common framework concerning methodological choices and assumptions for LCAs of electric vehicles.

Before assessing the environmental performances of specific vehicles, the most relevant aspects of performing a LCA of vehicles were identified and some recommendations for future LCA of vehicles were provided. The methodological aspects were identified according to the ILCD Handbook (EC - JRC, 2010a). The detailed description of the LCA steps is reported in APPENDIX A, while hereinafter the main conclusions and recommendations to perform LCA of vehicles are reported.

| |
|---|
| Complete LCA should be preferred to partial LCA in order to avoid the impacts shifting phenomena and to take into account the whole life-cycle of vehicles. In this perspective, auxiliaries' equipment should be part of the LCA of vehicles |

The environmental assessment of the fuel cycle should include both the fuel production and its consumption during the use-phase of vehicles (Stoner et al., 2007a, 2007b). In fact, in order to embrace a holistic perspective, several studies suggest to adopt a Well-to-Wheel (WtW) approach[46] (Chlopek and Lasocki, 2011; Messagie et al., 2014a; Nordelöf et al., 2014). Note that detailed rules for calculating the GHG emissions from renewable resources (i.e. biofuels) at the Well-to-Tank (WtT) stage are illustrated in Directive 2009/28/EC (EU, 2009).

Figure 22: Schematization of WtT, TtW, WtW and Equipment life-cycles (Nordelöf et al., 2014)




A "Complete Life Cycle Assessment", including all the life-cycle stages of a vehicle, also allow to include the impacts of the "Equipment life-cycle", i.e. the acquisition of raw materials and fuels, production stage, use-phase and EoL of such components (Figure 22). This will avoid the impacts shifting among different life-cycle stages and different environmental and human health problem fields (Messagie et al., 2014a) (Nemry et al., 2008) (Chlopek and Lasocki, 2011). The importance of these phases is also underlined by policy documents supporting eco-design of vehicles: this highlight that the reduction of the environmental impact of vehicles may be reached also through

[46] The WtW approach consider all the life-cycle phases of the vehicles, from the extraction of raw materials to their disposal This includes both the "well-to-tank" and the "tank-to-wheel" assessment". The first one consider the extraction the oil and transports (of the oil to the refinery as well as the distribution phase to the consumers). The second one takes into account the typology of fuel (biofuel, natural gas, conventional fuel), the exhaust gas treatment and the efficiency of the vehicle propulsion system.

the employment of new design options, like new materials, recycling/reusability improvements, etc. (EU, 2000).

In addition, the manufacturing of xEV is largely energy-intensive mainly due to the manufacturing of the battery production. Therefore, it is important to estimate the contribution of this life-cycle stage to the life-cycle impact since it affects not only the impact categories dominated by materials (e.g. ADP-res) but also impact categories dominated by energy (e.g. GWP) (EEA, 2018b) (Thomas et al., 2018).

| |
|---|
| Due to lack and incompleteness of data, assumptions and hypotheses become necessary and consequently the uncertainty of LCIA results increases. Therefore, specific LCI datasets and access to detailed inventories are needed also for permitting a representative comparison across LCIA of different vehicles. Moreover, iterative process is recommended. |

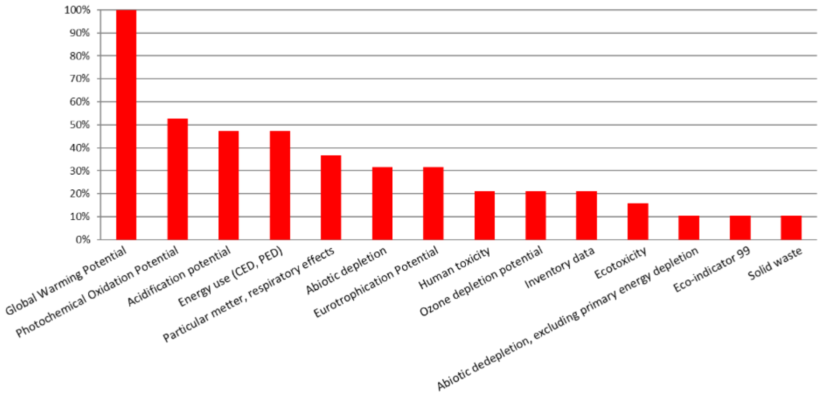
As previously mentioned, uncertainties are related to both the confidentiality of data and to the complexity of the value chain of vehicles. In fact, components hail from several suppliers all over the world and this gets more difficult to gather robust data for the LCI. Therefore, the data collected from the Automobile Industry departments (when available) are usually complemented by background databases such as Ecoinvent or Thinkstep databases to fill eventual gaps in collected data.

The iterativity of LCA allows identifying some specific aspects for which more in-depth analyses are recommended. Nonetheless, a strict collaboration with industrial partners could offer the access to primary (and more robust) data to model the LCI of vehicles.

| |
|---|
| In absence of specific methodologies for assessing the environmental burdens of vehicle, and for avoiding the impacts shifting, more than one impact category should be considered for the LCIA. |

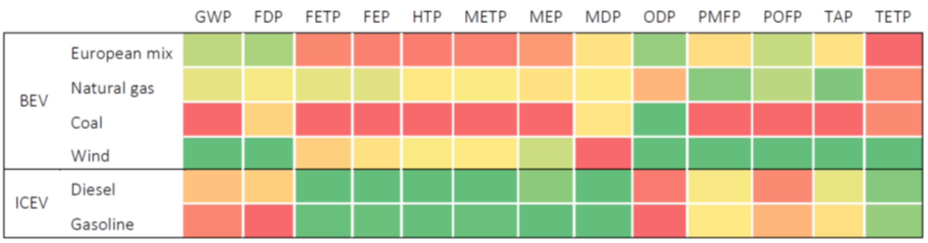
From the initial literature review performed on “LCA” of “ICEV” (internal combustion engine vehicle), it emerged that GWP is considered in all the addressed studies, while only 50% of them consider both POCP and CED, and 44% AP (Figure 23). Other impact categories are taken into account in few studies and they are not specifically developed for vehicle fleet, coherently with (Hawkins et al., 2012; Nordelöf et al., 2014) considerations. Note that most of the studies underline the need of considering resources depletion if xEV are assessed. Furthermore, some studies specifically address determined environmental aspects; for instance (Berger et al., 2012) evaluates the water footprint of EU cars through different characterization models (Ecological water scarcity, Motoshita et al., 2011 and Pfister et al., 2009). Huo et al. (2015) provide the LCA of electric vehicles in two different region of the world: China and US. It is significant that the “carbon intensity and cleanness of the electricity mix” of these two regions strongly affect the overall environmental burden.

Figure 23: Major impact categories adopted for the vehicles LCA in the performed literature review




The comparison between the life-cycle impacts of xEV and ICEV available in the literature confirm the importance of adopting a set of indicators instead of single or aggregated indicators. In fact, some impact categories that are not much relevant for ICEV can be highly relevant for xEV, e.g. those mainly affected by metal use as eutrophication and toxicity (Thomas et al., 2018) (EEA, 2018b) (Figure 24).

Figure 24: Life-cycle impacts of BEV and ICEVs (Thomas et al., 2018)




EoL of vehicle should be part of the system boundaries, also considering the development of recycling technologies of specific vehicles' components

Even for ICEV, the manufacturing and the EOL contribution to the overall impact is not negligible (Bobba et al., 2016b). Methodological aspects as allocation of debits/credits can affect the LCIA results but the performed study as well as the literature confirm that through the analysis of EoL, it is possible to identify some criticalities from an environmental point of view as well as potential improvements in the design phase. For instance, the aluminium content of a vehicle could be very relevant in terms of environmental credits.

In addition, the development of new EoL technology can affect the credits/debits of EoL but also the allocation of impacts. For instance, the most common recycling technology of traction batteries is a pyrometallurgical technology (Chagnes and Pospiech, 2013; Swain, 2017), which entails significant energy consumption and it is highly effective at recovering nickel, cobalt, copper and steel (Kushnir, 2015; Mancini et al., 2013). Aluminium, lithium and manganese are lost in the sludge since it is not economic or energy efficient to recover them; the slug can be used as an aggregate in concrete (Alves Dias et al., 2018; Boyden et al., 2014; Dunn et al., 2012). However, among the metallurgy processes, hydrometallurgical and combined processes are arising (Friedrich and Peters, 2017; Mathieux et al., 2017; Swain, 2017) in order to recover much more materials, e.g. *metal sulphate, which can be used again to manufacture batteries' active materials* (Recharge Association, 2018) (section 4.3.1.2).

## 3.2 Life Cycle Assessment (LCA) of passenger cars

In this section, the development of a modular LCA to assess different vehicles is illustrated. Primary information and the profound knowledge in the sector provided by CRF was used to develop the modular and parametrized LCA model. Then, the LCA was modelled according to the ISO14040 (ISO, 2006a) and ISO14044 (ISO, 2006b). In this chapter, the most important findings of the LCA are illustrated. Note that due to confidentiality of information, in the chapter, the steps of the performed analysis are reported but quantitative data cannot be disclosed.

All the LCA analyses were critical reviewed by certified auditors.

### 3.2.1 Goal and scope

The main goal of this LCA is the comparison between the life-cycle environmental performances of the four FCA models.

The case studies of the LCA activity are:

1. *A segment gasoline old* manufactured in Poland;
2. *A segment gasoline new* manufactured in Italy.
3. *B segment gasoline 1*;
4. *B segment gasoline 2*.

The Functional Unit (FU) of the study is the passenger vehicle for a lifetime of 200,000 km taking into account the New European Driving Cycle (NEDC) and respecting M1 type approval requirements and the other automotive legislations as well. According to the EU Directive 2009/33/EC, the functional units of the study are:

1. The *A segment gasoline old* with a use phase of 200,000 km;
2. The *A segment gasoline new* (gasoline fuel) with a use phase of 200,000 km.
3. The *B segment gasoline 1* (gasoline fuel) with a use phase of 200,000 km;
4. The *B segment gasoline 2* (gasoline fuel) with a use phase of 200,000 km.

Consistently with the literature review (section3.1), the performed analysis is a *from-cradle-to-grave* LCA. In the following, the vehicles life-cycle was schematized in three main phases:

- Manufacturing phase;
- Use phase (*well-to-wheel*);
- End-of-life.

Concerning the manufacturing, the most important data source is represented by International Material Data System (IMDS). Data are provided as aggregated materials in different classification group. CRF knowledge and contacts ease to identify correspondences within the LCA datasets. Moreover, assumptions about the LCI model are object of a following sensitivity analysis (section 0).

For the use phase, primary data provided by FCA group are used. The Standard Aggregation

Data (SAD)[47] is used for data about the environmental performances at plant level. Both homologation data (i.e. $CO_2$ emissions and fuel consumption) and legislative data (tailpipe emissions in accordance with the vehicle emission threshold) were used. The maintenance of the two vehicles was not considered, according to previous LCA studies performed by FCA and on literature, e.g. Messagie et al. (2014b)[48].

The EoL was modelled based on previous FCA studies repartitioning the material breakdown based on the recyclability and recoverability indexes for each vehicle (ISO 22628 standard and 2000/53/EC Directive). The energy consumptions of the ELV management processes (depollution, dismantling, shredding) have also been included. Where necessary, literature data were used for filling gaps.

The LCA software used for performing the analysis is Gabi[49] version 6, and Thinkstep is the main database[50]. When necessary, data were integrated by the Ecoinvent v2.2 database[51] and/or the literature review. In this case, the process units were realized *ad hoc*.

In coherence to the previous studies of the FCA group, the selected potential impact categories belong to the CML2001 method. Note that also the *Primary energy from non-renewable resources (net cal. value)* was included (Frischknecht et al., 2007). The Resource Depletion impacts have been subdivided into *Abiotic Depletion Potential, mineral resources* (Bobba et al., 2015) (Table 13).

Table 13: Impact categories considered in the study

| **Potential impact categories** | | **Unit of Measure** | **Level*** |
|---|---|---|---|
| CML2001 - Apr. 2013, Abiotic Depletion | ADP el | [kg Sb-Equiv.] | II |
| CML2001 - Apr. 2013, Acidification Potential | AP | [kg SO2-Equiv.] | II |
| CML2001 - Apr. 2013, Eutrophication Potential | EP | [kg Phosphate-Equiv.] | II |
| CML2001 - Apr. 2013, Global Warming Potential | GWP | [kg CO2-Equiv.] | I |
| CML2001 - Apr. 2013, Ozone Layer Depletion Potential | ODP | [kg R11-Equiv.] | I |
| CML2001 - Apr. 2013, Photochem. Ozone Creation Potential | POCP | [kg Ethene-Equiv.] | II |
| Primary energy from non-renewable resources (gross calorific value) | PED | [MJ] | n.a. |

* according to ILCD Handbook, levels specify the level of recommendation to use the correspondent impact categories (Level I means recommended and satisfactory, Level II means recommended but in need of some improvements, Level III means recommended, but to be applied with caution)

[47] http://2013interactivesustainabilityreport.fcagroup.com/en/processes/world-class-manufacturing-and-process-certification/organization-environmental-performance#start

[48] The environmental contribution of the car maintenance is low compared to the car life-cycle impacts. Moreover, in the Ecoinvent database, the maintenance impact of an average gasoline passenger car is lower than 4%, with exception for the abiotic depletion potential impact category. Furthermore, it is worthy that the maintenance is strictly depending on the driver behaviour, so that several scenarios should be considered, increasing the subjectivity of the analysis. Finally, it is assumed that the maintenance of these two vehicles is similar, thus it would not affect the comparison of their environmental performances

[49] http://www.gabi-software.com/italy/index/

[50] http://www.gabi-software.com/italy/databases/

[51] http://www.ecoinvent.org/database/database.html

To enlarge the analysis, also the ILCD/PEF recommended impact categories are used. LCIA results are illustrated in APPENDIX B.

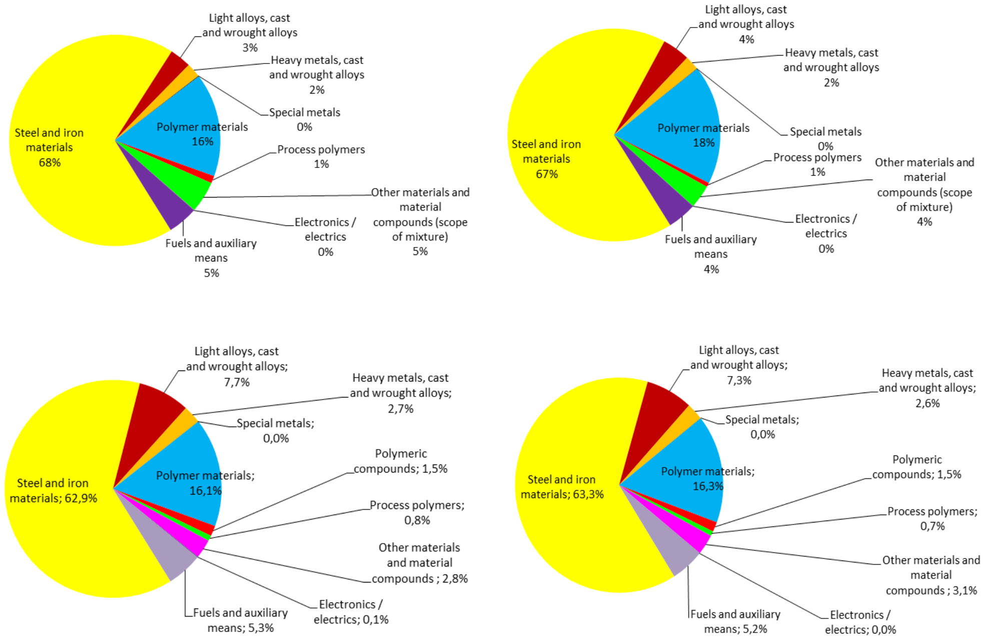
## 3.2.2 Life Cycle Inventory (LCI)

Due to the confidentiality of information, the detailed LCI is not reported in this document. However, the description of the life-cycle modelling is provided hereinafter.

**<u>Manufacturing</u>**

Figure 25 shows the most relevant material category of the assessed models. The most important material in terms of mass contribution is represented by the "Steel and iron materials" category, while all the other categories (except for the "Polymer materials") have a contribution lower than 5%. Note that the new model vehicle is heavier than the old one (of about 51 kg); this difference is mostly due to the "Steel / cast steel / sintered steel" (increase of about 43 kg compared to the old model) and the "filled thermoplastics" (increase of about 31 kg compared to the old model).

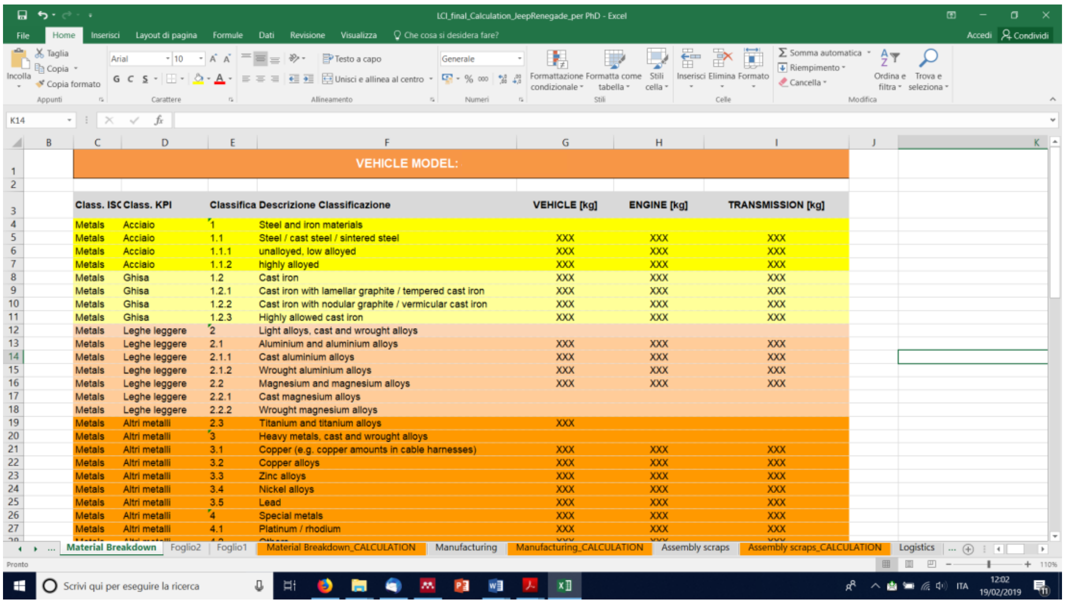
Figure 25: Materials pie chart of *A segment gasoline old (top left), A segment gasoline new (top right), B segment gasoline 1 (bottom left), B segment gasoline 2 (bottom right)*



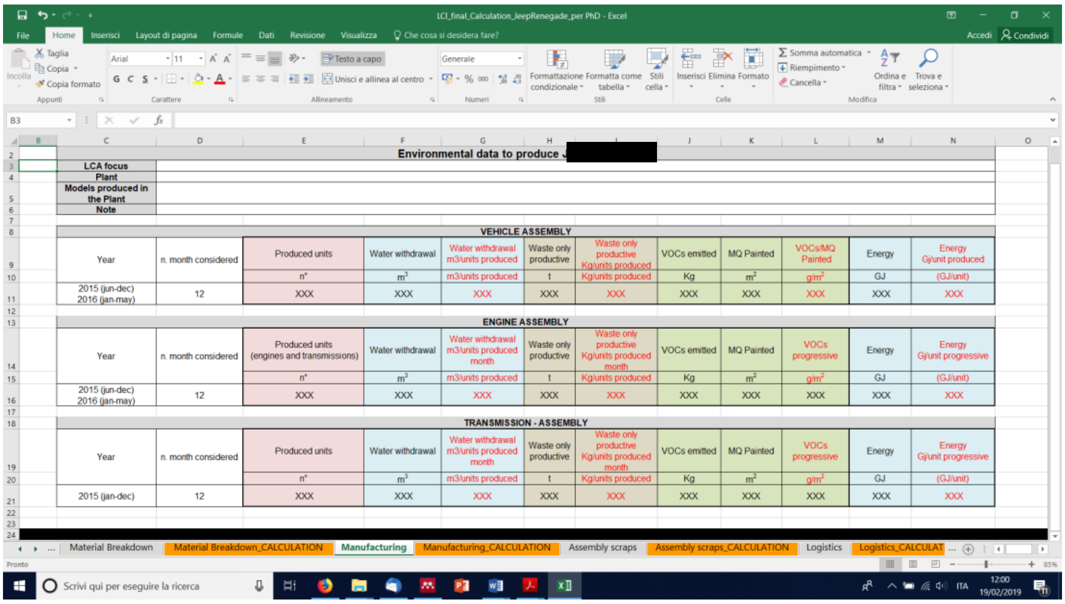

In the study, a special focus on engine and transmission components was provided due to their significant weight. Moreover, CRF have primary data on manufacturing data.

Concerning the vehicles' manufacturing, primary data about energy consumption, water withdrawals, waste management and emissions were collected directly from the manufacturing plants (based on the plants yearly information). The energy mix used in the LCA model was selected according to the geographical location of the manufacturing plants. Figure 26 gives an overview of the manufacturing information provided by CRF, based on the IMDS and primary data on vehicles assembly from FCA plants.

Figure 26: Primary data provided by FCA for all the assessed vehicles (materials and assembly information)



| Class. ISC | Class. KPI | Classifica | Descrizione Classificazione | VEHICLE [kg] | ENGINE [kg] | TRANSMISSION [kg] |
|---|---|---|---|---|---|---|
| VEHICLE MODEL: | | | | | | |
| Metals | Acciaio | 1 | Steel and iron materials | | | |
| Metals | Acciaio | 1.1 | Steel / cast steel / sintered steel | XXX | XXX | XXX |
| Metals | Acciaio | 1.1.1 | unalloyed, low alloyed | XXX | XXX | XXX |
| Metals | Acciaio | 1.1.2 | highly alloyed | XXX | XXX | XXX |
| Metals | Ghisa | 1.2 | Cast iron | XXX | XXX | XXX |
| Metals | Ghisa | 1.2.1 | Cast iron with lamellar graphite / tempered cast iron | XXX | XXX | XXX |
| Metals | Ghisa | 1.2.2 | Cast iron with nodular graphite / vermicular cast iron | XXX | XXX | XXX |
| Metals | Ghisa | 1.2.3 | Highly allowed cast iron | XXX | XXX | XXX |
| Metals | Leghe leggere | 2 | Light alloys, cast and wrought alloys | | | |
| Metals | Leghe leggere | 2.1 | Aluminium and aluminium alloys | XXX | XXX | XXX |
| Metals | Leghe leggere | 2.1.1 | Cast aluminium alloys | XXX | XXX | XXX |
| Metals | Leghe leggere | 2.1.2 | Wrought aluminium alloys | XXX | XXX | XXX |
| Metals | Leghe leggere | 2.2 | Magnesium and magnesium alloys | XXX | XXX | XXX |
| Metals | Leghe leggere | 2.2.1 | Cast magnesium alloys | | | |
| Metals | Leghe leggere | 2.2.2 | Wrought magnesium alloys | | | |
| Metals | Altri metalli | 2.3 | Titanium and titanium alloys | XXX | | |
| Metals | Altri metalli | 3 | Heavy metals, cast and wrought alloys | | | |
| Metals | Altri metalli | 3.1 | Copper (e.g. copper amounts in cable harnesses) | XXX | XXX | XXX |
| Metals | Altri metalli | 3.2 | Copper alloys | XXX | XXX | XXX |
| Metals | Altri metalli | 3.3 | Zinc alloys | XXX | XXX | XXX |
| Metals | Altri metalli | 3.4 | Nickel alloys | XXX | XXX | XXX |
| Metals | Altri metalli | 3.5 | Lead | XXX | XXX | XXX |
| Metals | Altri metalli | 4 | Special metals | XXX | XXX | XXX |
| Metals | Altri metalli | 4.1 | Platinum / rhodium | XXX | XXX | XXX |



Environmental data to produce J[redacted]

| | |
|---|---|
| LCA focus | |
| Plant | |
| Models produced in the Plant | |
| Note | |

VEHICLE ASSEMBLY

| Year | n. month considered | Produced units | Water withdrawal | Water withdrawal m3/units produced | Waste only productive | Waste only productive Kg/units produced | VOCs emitted | MQ Painted | VOCs/MQ Painted | Energy | Energy Gj/unit produced |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | n° | $m^3$ | m3/units produced | t | Kg/units produced | Kg | $m^2$ | $g/m^2$ | GJ | (GJ/unit) |
| 2015 (jun-dec) 2016 (jan-may) | 12 | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX |

ENGINE ASSEMBLY

| Year | n. month considered | Produced units (engines and transmissions) | Water withdrawal | Water withdrawal m3/units produced month | Waste only productive | Waste only productive Kg/units produced month | VOCs emitted | MQ Painted | VOCs progressive | Energy | Energy Gj/unit progressive |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | n° | $m^3$ | m3/units produced | t | Kg/units produced | Kg | $m^2$ | $g/m^2$ | GJ | (GJ/unit) |
| 2015 (jun-dec) 2016 (jan-may) | 12 | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX |

TRANSMISSION - ASSEMBLY

| Year | n. month considered | Produced units | Water withdrawal | Water withdrawal m3/units produced month | Waste only productive | Waste only productive Kg/units produced month | VOCs emitted | MQ Painted | VOCs progressive | Energy | Energy Gj/unit progressive |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | | n° | $m^3$ | m3/units produced | t | Kg/units produced | Kg | $m^2$ | $g/m^2$ | GJ | (GJ/unit) |
| 2015 (jan-dec) | 12 | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX | XXX |

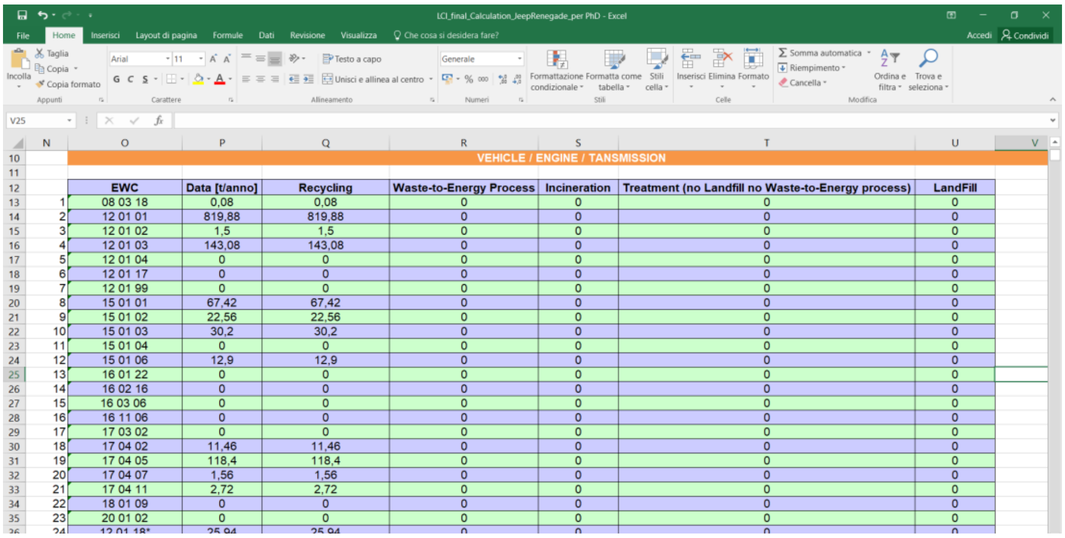
Waste generated during car manufacturing, excluding waste from facilities not directly involved in manufacturing, was assessed based on the European Waste Catalogue (EWC)[52] classification. The LCA model includes both the production and the EoL of the scraps arising from manufacturing (i.e. wastes from the assembly plants)[53]. Data provided by CRF are organised to be directly imported in the LCA software (Figure 27).

---

[52] http://ec.europa.eu/environment/waste/framework/list.htm

[53] In some cases, it was not possible to find an unambiguous correspondence between the EWC and a specific material (for instance, this is the case of sludge, e.g. 08 01 13*), hence some input materials were not included into the model.

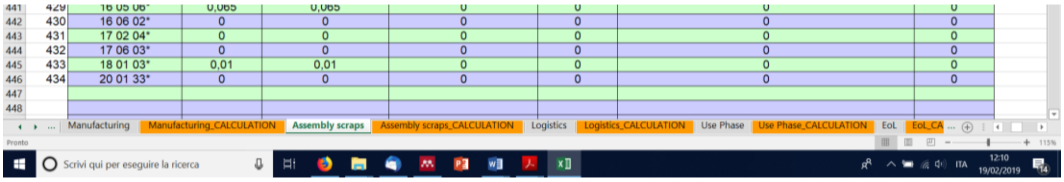
Figure 27: Primary data provided by FCA for all the assessed vehicles (materials and assembly information)



VEHICLE / ENGINE / TANSMISSION

| | EWC | Data [t/anno] | Recycling | Waste-to-Energy Process | Incineration | Treatment (no Landfill no Waste-to-Energy process) | LandFill |
|---|---|---|---|---|---|---|---|
| 1 | 08 03 18 | 0,08 | 0,08 | 0 | 0 | 0 | 0 |
| 2 | 12 01 01 | 819,88 | 819,88 | 0 | 0 | 0 | 0 |
| 3 | 12 01 02 | 1,5 | 1,5 | 0 | 0 | 0 | 0 |
| 4 | 12 01 03 | 143,08 | 143,08 | 0 | 0 | 0 | 0 |
| 5 | 12 01 04 | 0 | 0 | 0 | 0 | 0 | 0 |
| 6 | 12 01 17 | 0 | 0 | 0 | 0 | 0 | 0 |
| 7 | 12 01 99 | 0 | 0 | 0 | 0 | 0 | 0 |
| 8 | 15 01 01 | 67,42 | 67,42 | 0 | 0 | 0 | 0 |
| 9 | 15 01 02 | 22,56 | 22,56 | 0 | 0 | 0 | 0 |
| 10 | 15 01 03 | 30,2 | 30,2 | 0 | 0 | 0 | 0 |
| 11 | 15 01 04 | 0 | 0 | 0 | 0 | 0 | 0 |
| 12 | 15 01 06 | 12,9 | 12,9 | 0 | 0 | 0 | 0 |
| 13 | 16 01 22 | 0 | 0 | 0 | 0 | 0 | 0 |
| 14 | 16 02 16 | 0 | 0 | 0 | 0 | 0 | 0 |
| 15 | 16 03 06 | 0 | 0 | 0 | 0 | 0 | 0 |
| 16 | 16 11 06 | 0 | 0 | 0 | 0 | 0 | 0 |
| 17 | 17 03 02 | 0 | 0 | 0 | 0 | 0 | 0 |
| 18 | 17 04 02 | 11,46 | 11,46 | 0 | 0 | 0 | 0 |
| 19 | 17 04 05 | 118,4 | 118,4 | 0 | 0 | 0 | 0 |
| 20 | 17 04 07 | 1,56 | 1,56 | 0 | 0 | 0 | 0 |
| 21 | 17 04 11 | 2,72 | 2,72 | 0 | 0 | 0 | 0 |
| 22 | 18 01 09 | 0 | 0 | 0 | 0 | 0 | 0 |
| 23 | 20 01 02 | 0 | 0 | 0 | 0 | 0 | 0 |
| 24 | 12 01 18* | 25,94 | 25,94 | 0 | 0 | 0 | 0 |

…



| | EWC | Data [t/anno] | Recycling | Waste-to-Energy Process | Incineration | Treatment (no Landfill no Waste-to-Energy process) | LandFill |
|---|---|---|---|---|---|---|---|
| 429 | 16 05 06* | 0,065 | 0,065 | 0 | 0 | 0 | 0 |
| 430 | 16 06 02* | 0 | 0 | 0 | 0 | 0 | 0 |
| 431 | 17 02 04* | 0 | 0 | 0 | 0 | 0 | 0 |
| 432 | 17 06 03* | 0 | 0 | 0 | 0 | 0 | 0 |
| 433 | 18 01 03* | 0,01 | 0,01 | 0 | 0 | 0 | 0 |
| 434 | 20 01 33* | 0 | 0 | 0 | 0 | 0 | 0 |



## <u>Use phase</u>

The environmental contribution of the use-phase within the car life-cycle is mainly related to the fuel consumption and the direct emissions. According the EU Directive (2009/33/EC), the lifetime of the vehicles was considered equal to 200.000 km, which is the lifetime mileage of passenger cars as defined in Directive 2007/46/EC. Table 14 shows emissions and fuel consumption of two *A segment gasoline* vehicles assessed in this LCA. Data used in the evaluation refer to the NEDC driving cycle, which is the reference cycle for the vehicle type approval. In particular, the fuel consumption and the CO2 emissions have been evaluated using homologation data, while the other tailpipe emissions are referred to legislative thresholds on the basis of the environmental level of each vehicle (EURO 5 for the old model and EURO 6 for the new model).

Table 14: Use phase information for the four assessed vehicles

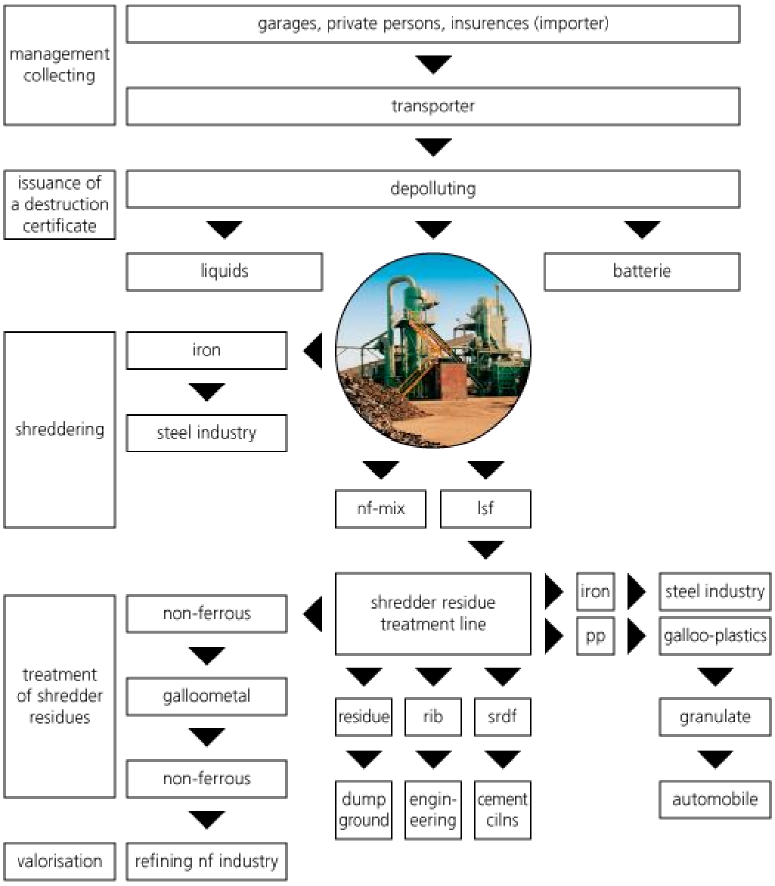
| **Description** | **Unit of measure** | **A segment gasoline old** | **A segment gasoline new** | **B segment gasoline 1** | **B segment gasoline 2** |
|---|---|---|---|---|---|
| CO2 | [g/km] | 113.00 | 110.00 | 140.00 | 139.00 |
| CO | [g/km] | 1.00 | 1.00 | 1.00 | 1.00 |
| HC | [g/km] | 0.10 | 0.10 | 0.10 | 0.10 |
| NOX | [g/km] | 0.06 | 0.06 | 0.06 | 0.06 |
| PM | [g/km] | 0.005 | 0.005 | 0.005 | 0.005 |
| Gasoline | [l/100 km] | 4.9 | 4.7 | 7.6 – urban<br>5.1 – suburban<br>6.0 average | 7.8 – urban<br>5.0 – suburban<br>6.0 average |

## End-of-Life (EoL)

To model the EoL of vehicles, three different steps were considered

- Depollution (or "Pre-treatment");
- Dismantling;
- Shredding.

Figure 28: Example of cars EoL treatments (http://www.galloo.com/node/5?language=en#sub1)




Depending on specific materials, both reuse, recycling, energy recovery and landfilling processes were considered after each of the above-mentioned steps. Note that all materials entering in the "Pre-treatment" phase are considered as recycled or reused based on FCA available information. Material fractions for each steps are reported in Table 15.

Table 15: materials fraction after for each EOL step (Bobba et al., 2016b)

| | Treatment after depollution | | | Treatment after Shredding | | | Landfill |
|---|---|---|---|---|---|---|---|
| | Reuse [%] | Recycling [%] | Energy Recovery [%] | Reuse [%] | Recycling [%] | Energy Recovery [%] | [%] |
| **Metals** | 0 – 7 | --- | --- | --- | 80 - 95 | --- | 0 - 2 |
| **Plastics** | --- | 10 - 25 | --- | --- | 5 - 10 | 50 - 65 | 10 - 20 |
| **Glass** | --- | 0 - 5 | --- | --- | 80 - 95 | --- | 5 - 10 |
| **Fluids** | 30 - 60 | 30 - 50 | 30 - 50 | --- | --- | --- | --- |
| **Other** | 0 – 5 | --- | --- | --- | 5 - 15 | --- | 80 - 100 |

Credits for recycling are based on economic considerations considering the price ratio between the material scrap and the virgin one (Table 16).

Table 16: EoL information for the recycling processes

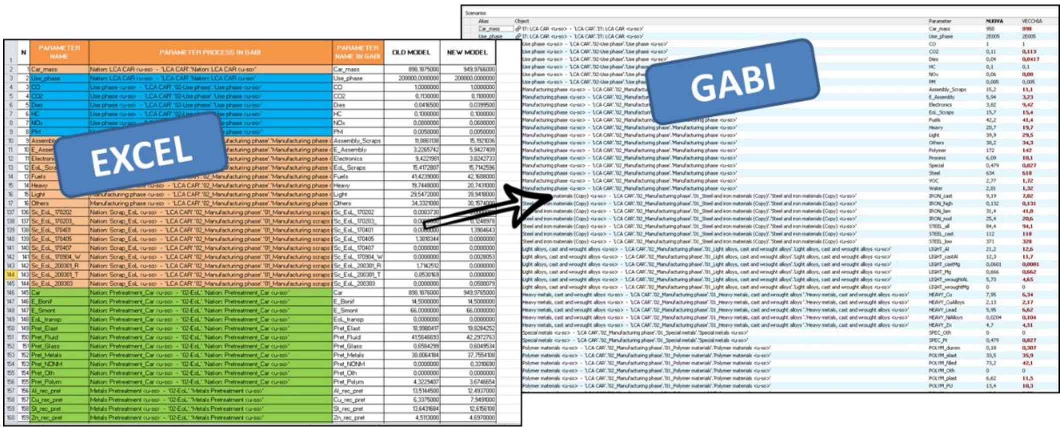
| Material | Energy for primary production [MJ/kg] | Energy for recycling process [MJ/kg] | Price of virgin material [€/ton] | Price of shredded material [€/ton] | Source<br>- Energy<br>- Prices virgin material<br>- Price shredded material |
|---|---|---|---|---|---|
| Aluminium | 279.00 | PE database | 1,530.21 | 1,330.93 | - PE database<br>- (COMEX - CME Group, 2015)<br>- (RIM, 2015) |
| Copper | 103.30 | 10.55 | 4,281.82 | 4,500.00 | - Bureau of International recycling (last accessed December 2015)<br>- (InfoMine, 2015a)<br>-(Mathews et al., 2015a) |
| Steel | 19.00 | 10.00 | 1,500.00 | 150.00-300.00 | - PE database<br>- (Fastmarkets, 2015)<br>- (EUROFER - the European Steel Association, 2015) |
| Zinc | 45.00 | 8.81 | 1,466.91 | 750.00 | - Bureau of International recycling (last accessed December 2015); (Zanotti and Scolari, 2012) /<br>-(InfoMine, 2015b)<br>- (Mathews et al., 2015b) |
| Elastomers | 95.00 | PE database | 2,258.00 | 300.00 | - PE database<br>- (Bundesverband Sekundärrohstoffe und Entsorgung, 2015)<br>- (Bundesverband Sekundärrohstoffe und Entsorgung, 2015) |
| Polypropylene | 79.00 | FCA data | 1,537.50 | 590.00 | - FCA data<br>- FCA data<br>- FCA data |
| Glass | 15.00 | 10.80 | 250.00 | 50.00 | - (Deflorian, 2008)<br>- FCA data<br>- (Eminton et al., 2015) |

## Life-cycle model

In order to create the life-cycle model, a strict connection between the data collection system and the GaBi software was created through the parametrization of the life-cycle model. This was based on a standardized data gathering (already performed in the FCA group). Therefore, a single GaBi model was adopted to provide different LCA analyses and potentially accelerate eventually future LCA analyses for other vehicles (i.e. comparable functional units). The correlation between inventory data and GaBi model was based on a correspondence between parameters used in the software and their definition deriving from the data collection (Figure 29).

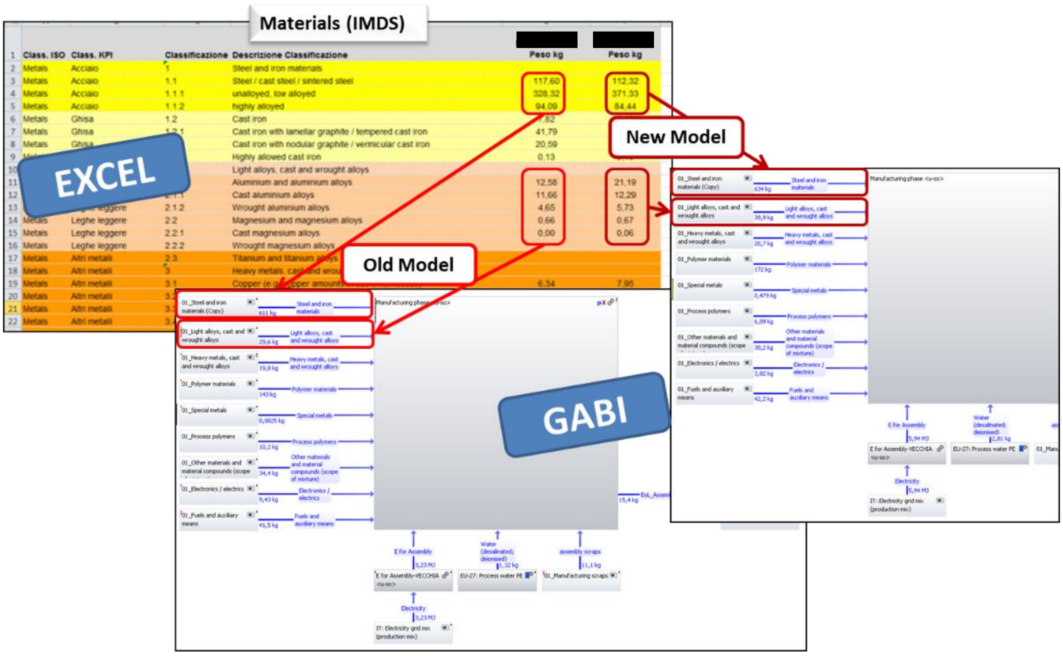
The modelling reflected the organization as well as the availability and the structure of the data collection already performed by FCA for each of the life-cycle phases of the assessed vehicles (Figure 30). When necessary, “sub-steps” were created, e.g. the manufacturing step was organised by three different sub-steps: materials for fabricating the vehicle components (one sub-step for each IMDS category), manufacturing scraps and manufacturing process (i.e. energy, direct emissions, etc.). Same approach was used for the use phase and the EoL modelling.

Figure 29: Example of the correlation between the FCA data collection and the GaBi parameters




For each of these steps, data were collected and parameters were associated to each specific process, depending on its characteristics. For instance, concerning the manufacturing phase, the parametrization pertained the amount of specific materials used for the car manufacturing (e.g. [kg] of steel) or scraps production during the manufacturing processes (e.g. [g] of VOCs emitted for the production of one car, [kg] of manufacturing scraps per produced vehicle).

Figure 30: Example of the correlation between the LCI and the GaBi model: for each IMDS category (from excel file) a GaBi plan was modelled



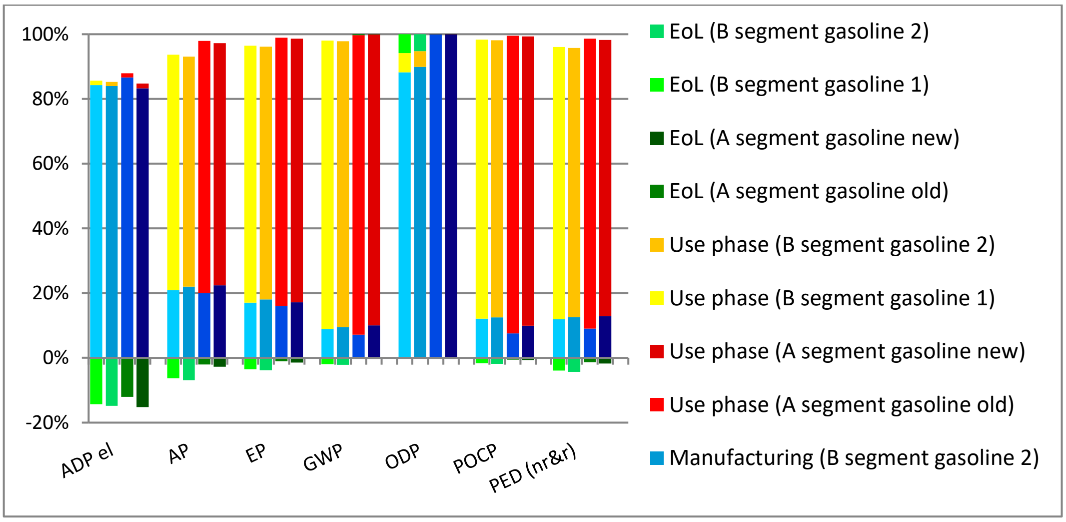

The added value of the model parametrization is substantially the potential development of future LCA analysis of FCA vehicles based on the same data collection structure and a single life-cycle model. Moreover, the use of more than 2,500 parameters could ease the accomplishment of sensitivity analyses.

## 3.2.3 Life Cycle Impact Assessment (LCIA)

All the potential impacts hereinafter illustrated have been calculated for the considered functional units. Figure 31 reports the contribution of the different life-cycle phases (manufacturing, use -for a lifetime of 200.000 km- and end-of-life) to the overall potential environmental impact.

Results show that the major contribution in terms of potential environmental impacts is associated to the use phase for almost all the assessed potential impact categories. The manufacturing phase is particularly significant for the ADP and the ODP impact categories. With an exception for the ODP indicator, all the other potential environmental impact categories highlight an environmental credit associated to the EoL, meaning that there are some environmental benefits related to the EoL treatments (e.g. reuse, recycling and incineration processes). It is to be noticed that the ADP impact category shows a remarkable contribution to the EoL of the vehicles, always higher than the threshold of -15% for all the models considered.

Figure 31: Percentage contribution of the life-cycle phases to the overall impact of the FCA assessed vehicles
(100% = life-cycle impact of the vehicle)



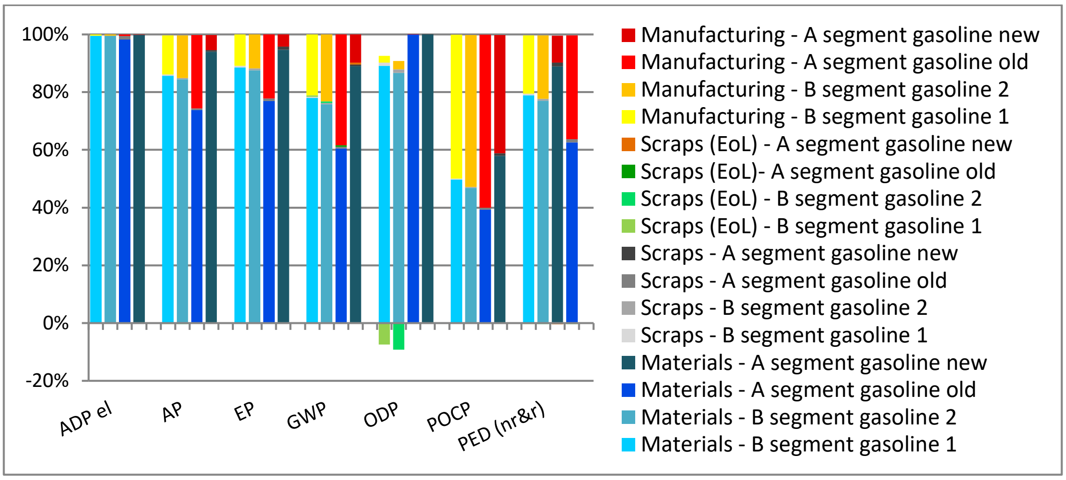

## 3.2.4 Life Cycle Interpretation

According to ISO 14040/44, in this chapter an in-depth analysis contribution for each of the life-cycle vehicle phases is illustrated.

**Manufacturing phase**

In order to highlight the relative contribution of the manufacturing phase inputs, materials used for the vehicle components, manufacturing scraps, energy, water withdrawal and direct emissions were grouped. The “scraps” include the contribution of the production of materials used during the manufacturing process (defined through the EWC, as illustrated in section 0) and the treatment of the scraps (recovery, recycling and landfill disposal). The manufacturing scraps (both their production and their EoL) as well as energy and water could be considered as negligible (Figure 32). Note that emissions occurring during the assembly process are important only for the POCP impact category (higher than 36% for all the vehicles’ models). The energy consumption associated to the manufacturing process (i.e. the assembly of motor, transmission and the whole vehicle) ranges between 1% and 36% of the manufacturing phase potential impact. Note that the manufacturing process includes energy, water and VOC and that, with exceptions for the ADP and the ODP impact categories, it shows always a contribution higher than 13%.

Figure 32: Contribution of the manufacturing processes to the vehicle manufacturing impact (manufacturing process includes both the energy, water and VOC painting) (100% = vehicle manufacturing impact)




Among materials used for the vehicle components, a contribution analysis was performed per materials category, according to IMDS:

1. Steel and iron materials
2. Light alloys, cast and wrought alloys
3. Heavy metals, cast and wrought alloys
4. Special metals
5. Polymer materials
6. Process polymers
7. Other materials and material compounds (scope of mixture)
8. Electronics / electrics
9. Fuels and auxiliary means

The contribution analysis highlights that, even if most important materials category in terms of mass contribution is “steel and iron materials” for all the vehicles’ models (about 60%65% of the total mass), it does not represent the most important materials category in terms of life-cycle potential impact for all the assessed potential impact categories (in fact its environmental contribution to the life-cycle potential impact is always lower than 40%, with the highest value for GWP).

On the contrary, despite the low weight contribution of the “light alloys, cast and wrought alloys” category to the vehicles mass (ranging between 3.29% and 7.73% according to the vehicle), this materials category assumes high potential impacts for some of the assessed potential impact categories. This is the case of GWP (respectively 15.61% and 21.83% for the old and the new *A segment gasoline* models) and of PED (respectively 6.02% and 22.33% for the *B segment gasoline 2*and the *A segment gasoline new*).

Note that, concerning “fuels and auxiliary means”, the LCIA of the 4 models show significant differences for the ODP impact category. This is mainly related to the R134a refrigerant. In fact, the LCI of both the *B segment gasoline 1* and *2* does not entail the refrigerant in the LCI. For the two *A segment gasoline* models, even if the contribution of the “fuels and auxiliary means” in mass is low (less than 5% for both the *A segment gasoline* models), their potential impact on the manufacturing phase is particularly relevant for ODP (higher than 99.89% for both the vehicles’ models).

Similarly, the “electronics / electrics” material category is particularly remarkable for ADP as well as the “heavy metals, cast and wrought alloys” materials category. In this case, the most important materials in terms of environmental burden are lead and zinc.

Finally, it is to be noticed that the negative contribution associated to the “Other materials and material compounds” refer to the “float glass PE” process unit and it strictly depends on the PE process unit modelling.

Figure 33: Contribution of the *B segment gasoline 1* and *2* vehicles materials to the vehicle manufacturing impact

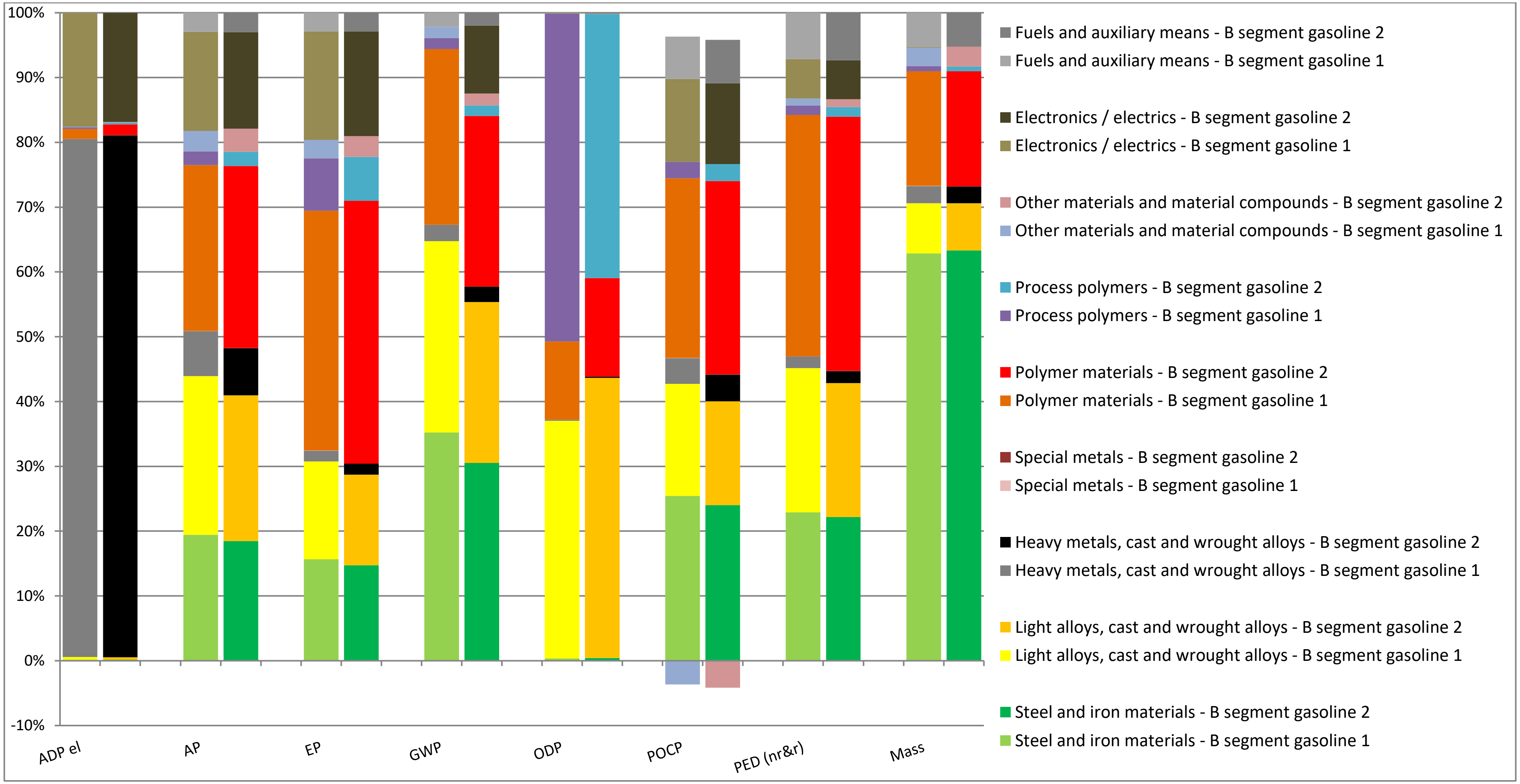
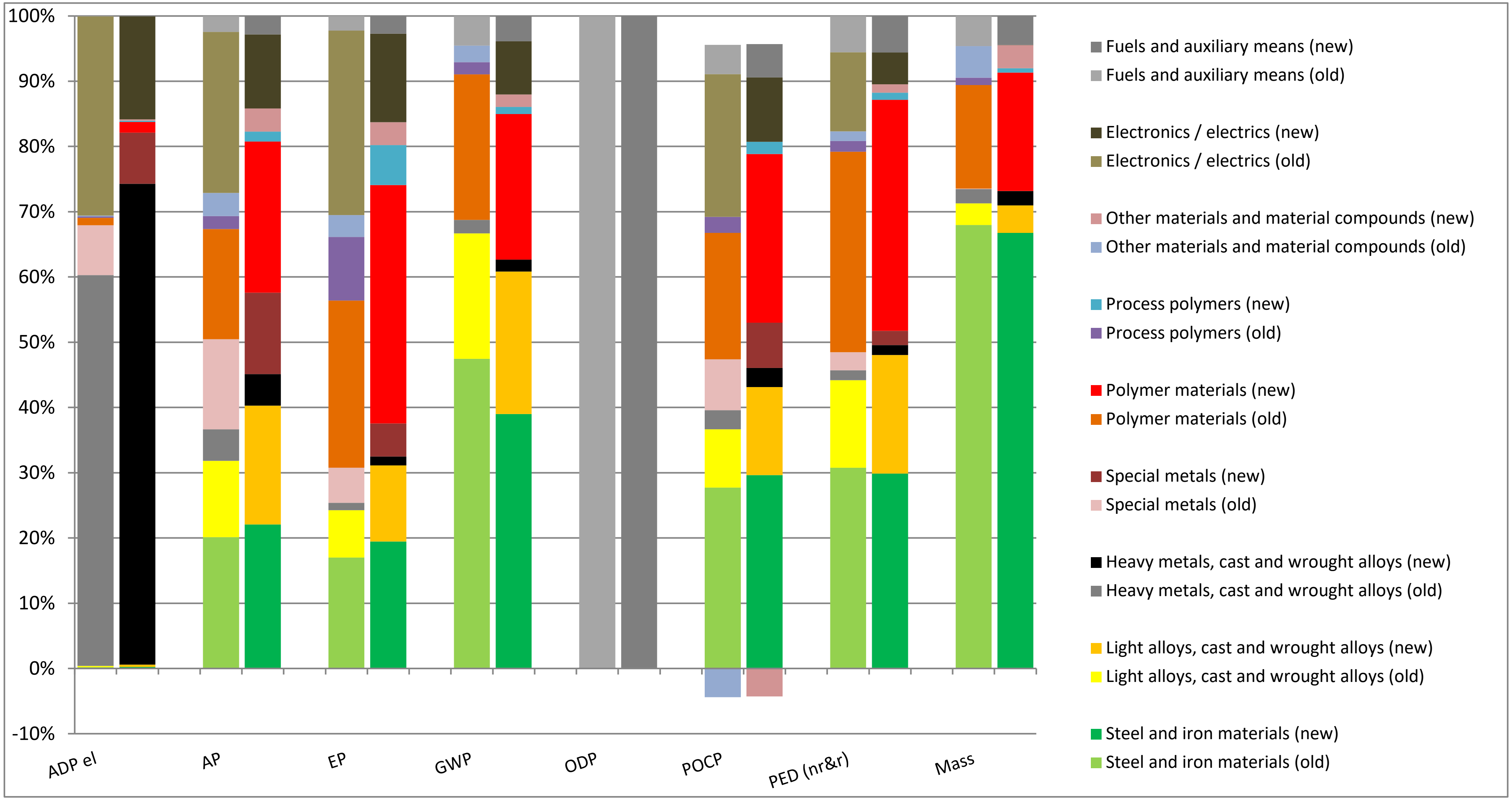
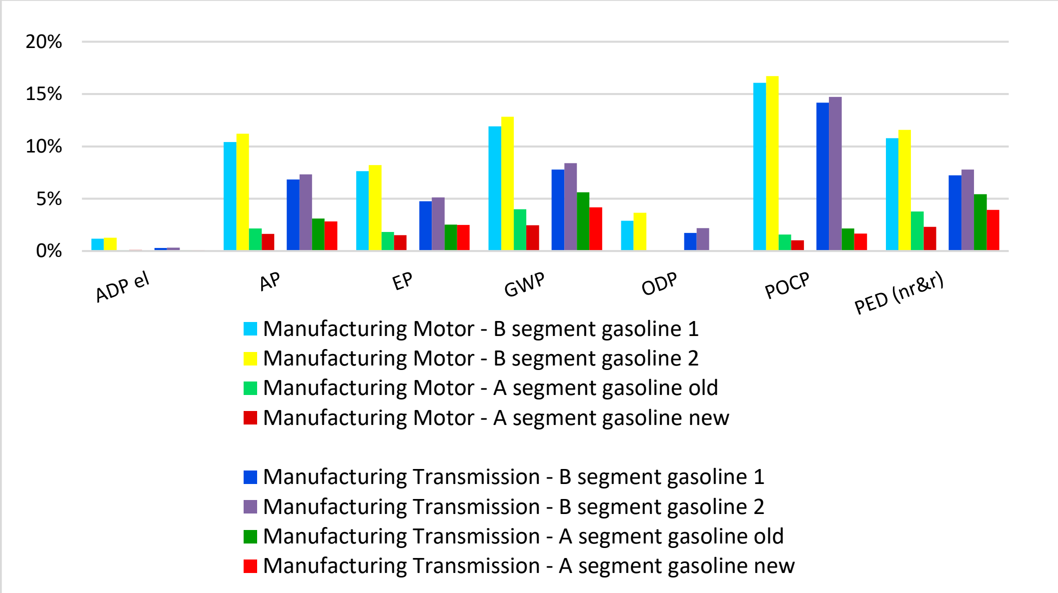


Figure 34: Contribution of the *A segment gasoline* vehicles materials to the vehicle manufacturing impact (100% = manufacturing impact of the old model)



The contribution of logistics operations of the vehicles main components (i.e. engine and transmission) may be considered as negligible if compared to the whole manufacturing potential impact (about 1% of the life-cycle impact). Moreover, results of the performed contribution analysis show that, with exception for the POCP impact category, the highest contribution to the manufacturing environmental impact is associated to the materials for both the engine and the transmission.

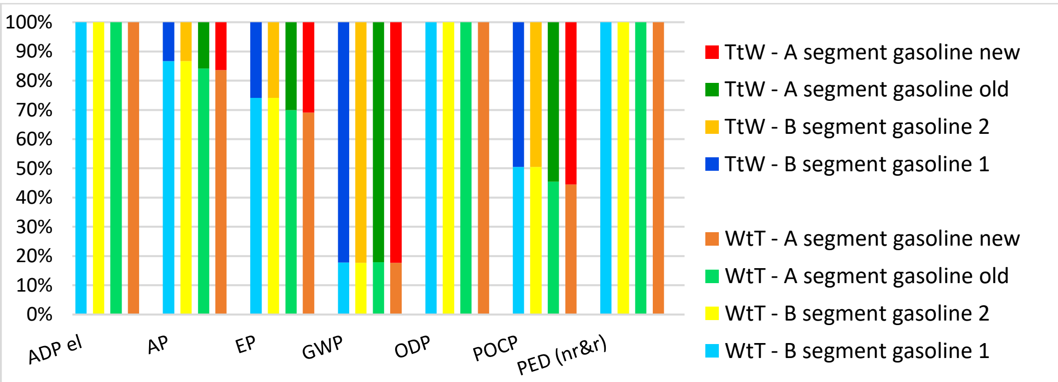
Figure 35: Contribution of the engine and transmission to the vehicle manufacturing impact (100% = vehicle manufacturing impact)




## Use Phase

The use phase environmental potential impact is affected by both the production of gasoline (*Well-to-Tank*) and the emissions during the use of the vehicles (*Tank-to-Wheel*). For all the models, some of the potential impact categories are mostly affected by the vehicle usage, such as GWP (mainly due to carbon dioxide emissions) and POCP (mainly due to the carbon monoxide and hydrocarbons emissions), while the others are mainly affected by gasoline production.

Figure 36: Contribution of gasoline (WtT) and emissions (TtW) to the vehicle use phase impact (100% = vehicle use phase impact)

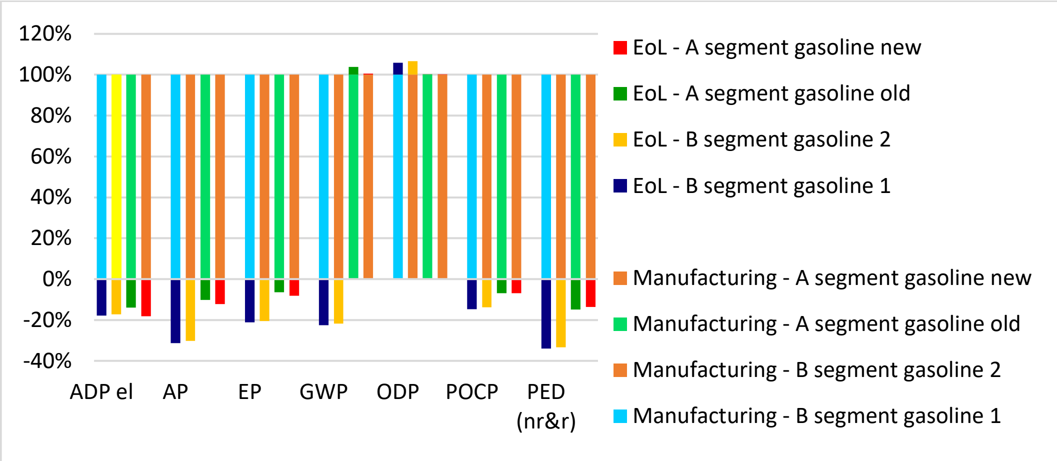


## EoL

The analysis shows that EoL can entail some environmental benefits. In fact, 6 out of 7 potential impact categories show that net gains in the EoL are higher than generated potential impacts and that the highest net credits refer to PED. In case of ODP, some of the EoL processes generate important potential impacts (e.g. fugitive emission of R134a).

Note that, as illustrated in section 0, the energy consumption for the recycling refer to the treatment of 100% of the material scrap, whereas the conversion factor was adopted only for the calculation of the environmental credits.

Figure 37: Ratio between the EoL impact to the manufacturing impact
(100% = vehicle manufacturing impact)




For all the EoL phases, a contribution analysis of specific materials categories was performed. Figure 38 gives an overview of the performed analysis. Detailed results are illustrated in Appendix C, while hereinafter the main outcomes are reported.

IN general, the in-depth analysis highlights that "recycling after shredding" and "recycling after depollution" have high net environmental credits; this latter represents almost all the benefit for the ADP impact category. Benefits associated to the incineration process are relatively low for all the potential impact categories and they represent an important potential impact for the GWP, mainly due to the plastics incineration. The energy contribution for the depollution, disassembly and shredding processes is low; it is worth to underline that the depollution phase is especially relevant for GWP mainly due to the R134a refrigerant (in the two *A segment gasoline* models[54]). Finally, the landfill disposal represents negligible environmental burdens for all the assessed impact category (highest contribution refers to EP).

Focusing on the materials contribution the contribution analysis highlights that, with exception for ADP (where bauxite is associated to very abundant reserves), all the other assessed potential impact categories highlight a remarkable contribution of aluminium: the recycling process of aluminium is to be associated to significant environmental credits. Also, important

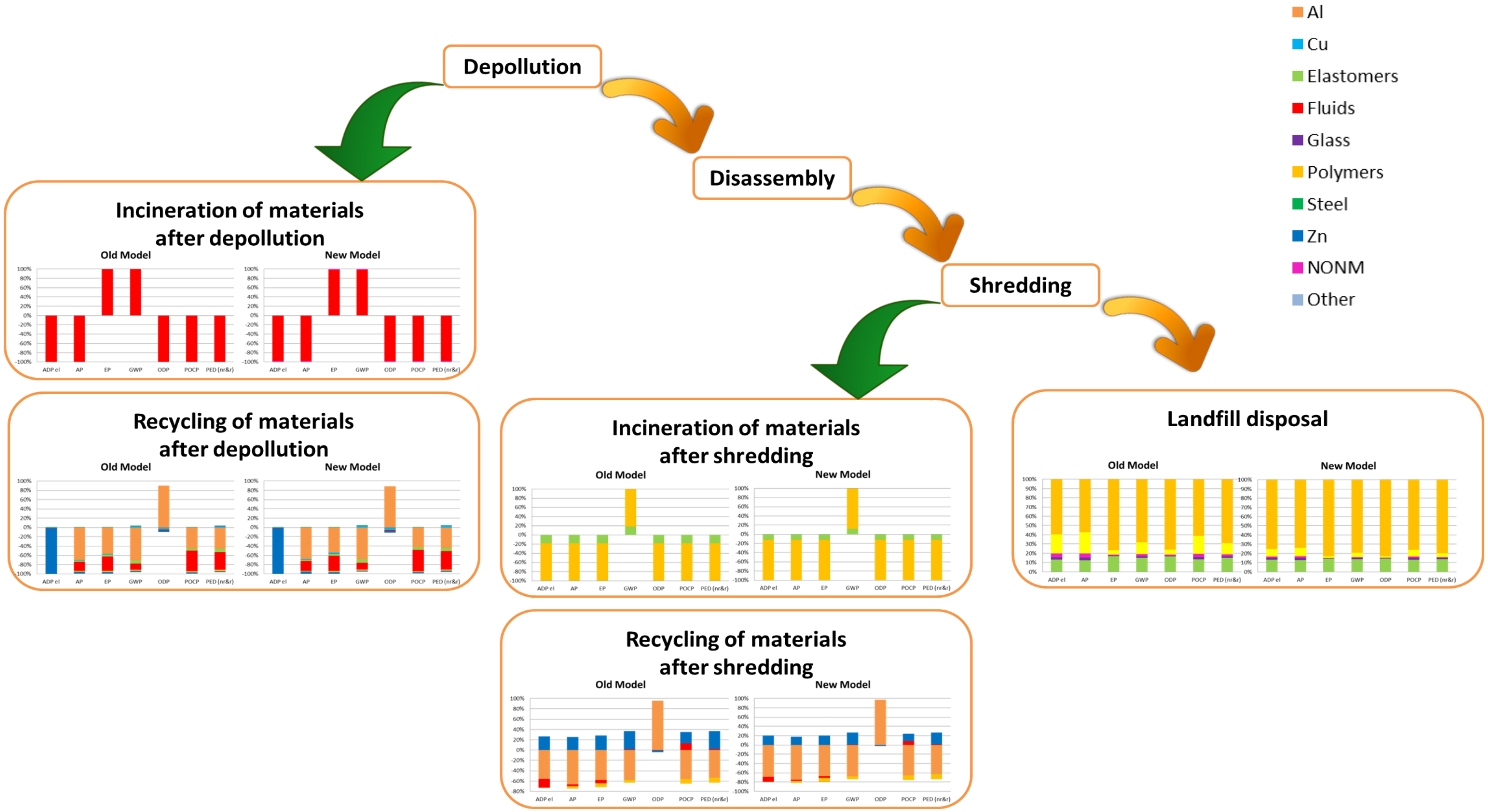
[54] Note that the LCI of both the B segment gasoline 1 and B segment gasoline 2 do not entail the refrigerant in the LCI

environmental credits are represented by the "fluid reuse", the "steel recycling" and the "polymer incineration. Zinc represents the most important contribution for the ADP impact category, due to relatively lower reserves in respect to other abiotic raw materials.

Finally, due to the high level of uncertainty and variability of the market price, both the Value-Corrected Substitution (VCS) approach (Johnson et al., 2013; Nicholson, 2009; Spielmann and Althaus, 2007) and the recycled content (RC) approach Johnson et al. (2013) were adopted. The VCS approach consider the ratio between the price of the virgin material and the price of the recycled materials w for the calculation of allocation factor; then, this value is multiplied by the amount of the avoided product in product manufacturing process. On the other hand, the RC approach assumes that all the reused and the recycled materials substitute the respective virgin materials (Bobba et al., 2016b). Lower environmental benefits are observed in case of adopting the VCS approach, especially for the ADP-res. However, in case of the *A segment gasoline* models, the difference between the two approaches is not so relevant in a life-cycle perspective, except for the ADP impact category[55]

[55] In this case, the life-cycle impact by the adoption of the VCS allocation approach is 1.18 and 1.26 times (respectively for the old and the new model) the life-cycle impact by the adoption of the RC allocation approach

Figure 38: Contribution of the materials treatments for each EoL phase



## 3.3 Relevant aspects of vehicles development for further studies

To become leader in a “clean, competitive and connected mobility” (EC, 2017b), the EU mobility requires a profound change. Focusing on the decarbonization of Europe, e-mobility can play a relevant role and the penetration rate of electric vehicles (xEV) in EU is increasing, already. The developed modular LCA could be used to assess the performance of different types of vehicles and, in case, consistently compare the obtained results. Thence, the main differences in terms of components between ICEVs and xEV should be identify in order to add and/or replace some LCA modules in the developed modular LCA. Focusing on the different units composing both the ICEVs and the xEVs, the macro-units are represented by the glider and the powertrain. Since the main difference between ICEVs and xEVs is the powertrain, the modules related to the glider of ICEVs could be used also for modelling the impacts of the glider of xEVs (Hawkins et al., 2013; Lettieri et al., 2015; Notter et al., 2010); indeed, the modules of the powertrain should be replaced by a new module modelling the electric powertrain, i.e. the electric motor and traction battery. Also, it is to be noticed that the module of EoL of the vehicles should be revised adding specific materials and processes related to the EoL treatment of the electric powertrain.

In the following, some relevant considerations to be considered in further work on LCA of xEV are reported. According to Notter et al. (2010) and Hawkins et al. (2013), the differences between the environmental impacts of the drive-trains of the xEV and the ICEVs are negligible. Therefore, this section will mainly focus on the traction battery.

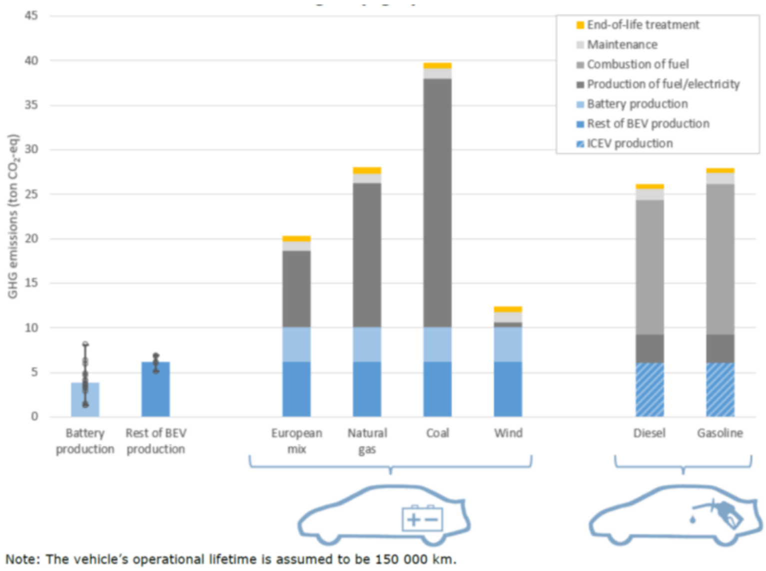
The electric power train is objective of several research activities, where a strong attention is addressed especially to batteries. This is also supported by the creation of the European Battery Alliance (European Commission, 2018) and the adoption of the Strategic Action Plan on Batteries in May 2018 as part of the third ‘Europe on the Move’ mobility package (EC, 2018b). The development of a sustainable battery value chain, which also means a battery production process with the lowest impacts possible, is recognised as a key driver for EU competitiveness and it is attracting a lot of interest.

From an environmental perspective, this transition from ICEV to e-mobility entails a high relevance of the environmental impacts related to manufacturing. In fact, while for ICEV the life-cycle impacts are mainly dominated by the use phase, for xEV, the contribution of the manufacturing of the new components (e.g. battery and electric motor) are quite relevant to the life-cycle impacts (Figure 39) (Rotter, 2017; Thomas et al., 2018). Moreover, this transition also require the exploitations of different materials compared to the materials needed for ICEV, e.g. large amount of lithium, manganese, cobalt, natural graphite etc. This is very relevant for Europe since some of the above mentioned materials belong to the list of CRMs for the EU (BOX 3) (Rotter, 2017).

Studies assessing both ICEV and xEV are available in the scientific literature, even though often their comparison is complex due to different assumptions and system boundaries. The contribution of the battery production to the life-cycle GHG is recognised as significant by several studies (Cusenza et al., 2019; Nordelöf et al., 2014; Thomas et al., 2018; Zackrisson et al., 2010), but it is also highlighted that comparison often refer to GHG Figure 39). According to the LCA methodology, a broader set of indicators is needed to have a more complete overview of the life-cycle environmental impacts of a product, especially when assessed products are complex.

Moreover, confidentiality of quantitative information make difficult to disaggregate data in order to reproduce the analyses.

Figure 39: Life-cycle GHG emissions of mid-sized 24 kWh battery electric (left) and internal combustion engine (right) vehicles (Thomas et al., 2018)




Observed discrepancies in available impact assessments' results are related to a set of interconnected factors. In the following, some significant aspects to perform LCAs of e-mobility are reported. Data refer to both available LCA studies in the literature and to relevant aspects related to the future development of the xEV market.

First of all, several types of batteries are already available in the market, i.e. different battery chemistry and different dimensions. These characteristics of the batteries are directly related to the use of raw materials and specific components for their manufacturing, e.g. binder. Among the chemistries available in the market (Figure 40), the most suitable LIBs chemistries for traction are NMC and NCA. The NMC batteries are expected to remain the larger share of traction LIB in the next years, even though the amount of embedded materials is already changing (e.g. decreasing amount of cobalt)[56] (Alves Dias et al., 2018; Lebedeva et al., 2016; Zubi et al., 2018).

The battery chemistry and dimensions influence other relevant variables: the capacity, the energy/power density, the number of full cycles and the cost. All these parameters affect the performance of the battery and therefore the distance that a xEV is able to cover with a specific battery (for instance, see Figure 41 and Figure 42).

[56] For instance, new chemistries with lower Co content are available already, e.g. NMC 523, 622, and 811 instead of NMC 111 (Berman et al., 2018; IEA, 2018; Perks, 2016; Pillot, 2017); also, the use of composite cathodes is another strategy to decrease the Co content (Cusenza et al., 2019; Patry et al., 2014).

Figure 40. Main characteristics of Li-ion batteries (Zubi et al., 2018)


Figure 41. Statistical relation between Leaf batteries' capacity and miles driven (the graph refers to the Nissan Leaf battery)

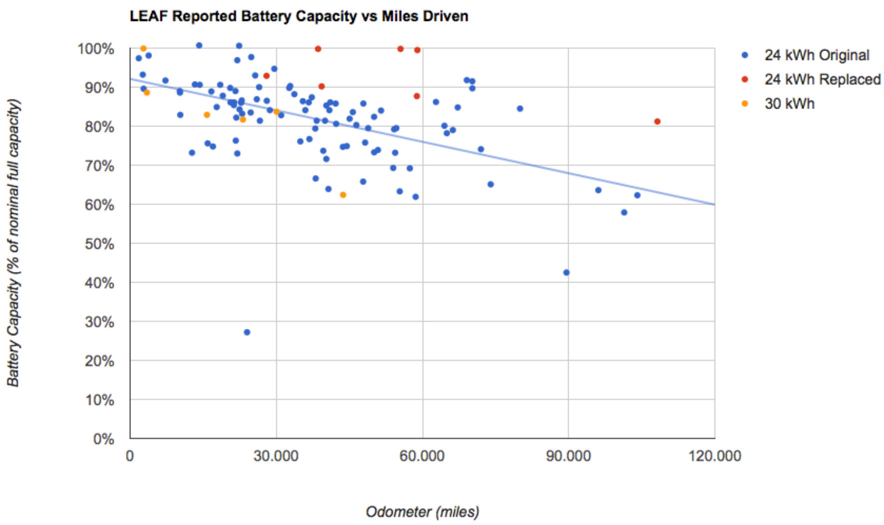
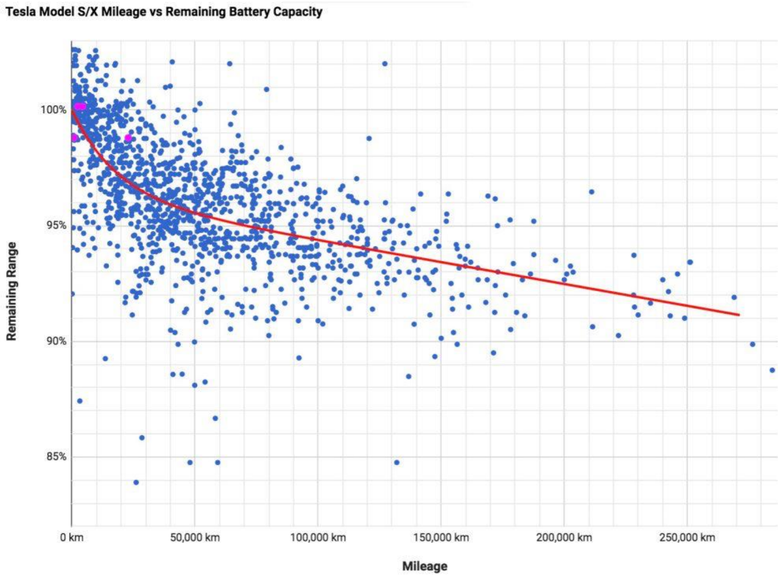


Figure 42. Statistical relation between Tesla batteries' capacity and kilometres driven (https://electrek.co/2018/04/14/tesla-battery-degradation-data/[57])



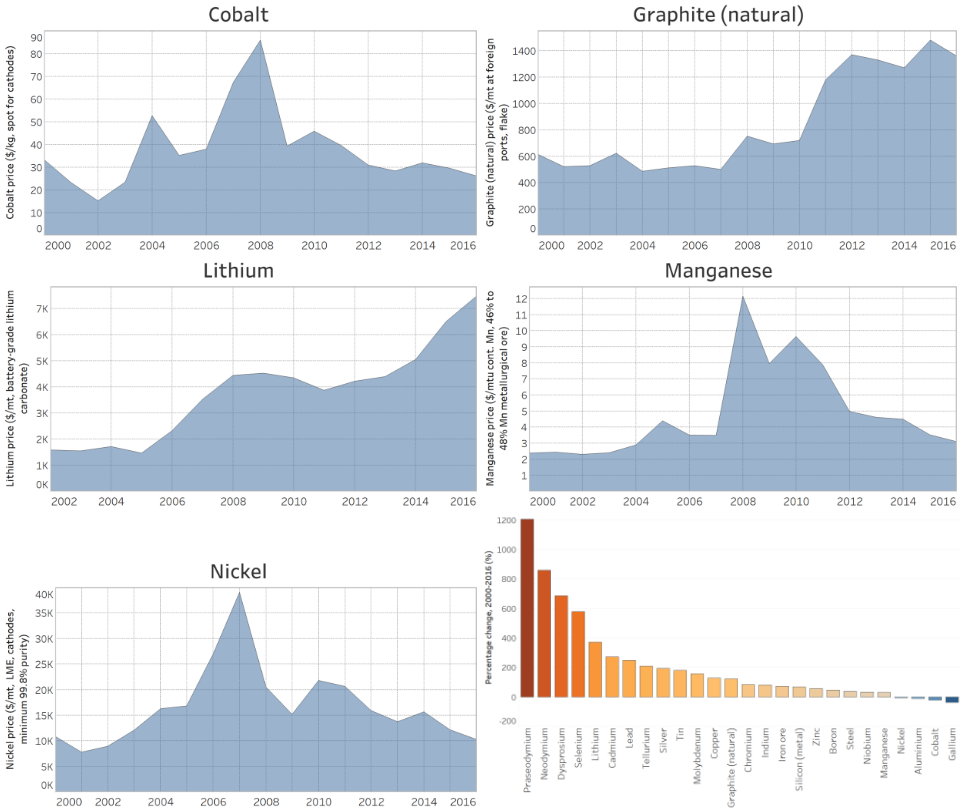

The price of LIBs mainly depends on the cells and therefore on the materials used in the cathodes. The price of both lithium and cobalt almost doubled in the first semester of 2017 due to the increasing popularity of EV. If a wider period is considered, price of cobalt did not significantly change between 2000 and 2016, as well as the price of manganese and nickel (Figure 43), while the price of both lithium and natural graphite increased significantly (Pavel and Blagoeva, 2017). Cobalt used for LIBs cathode is the most relevant cost item of the battery, representing about 3-6% of the cost of the NMC and NCA cells[58] (Zubi et al., 2018). Compared to lithium, changes in price of cobalt will affect the price of the battery much more than changes in price of lithium[59] (Zubi et al., 2018).

[57] Tesla survey can be downloaded from the web site: https://docs.google.com/spreadsheets/d/t024bMoRiDPIDialGnuKPsg/edit#gid=1710185683

[58] Considering that the cells represents about 65% of the battery cost

[59] The cost share of lithium in LIB's cell is around 1.2%

Figure 43. Raw materials prices from 2000 to 2016 (Pavel and Blagoeva, 2017)




A discrepancy emerging from published LCA studies is related to system boundaries. In facts, different LCA studies are not easily comparable since the system boundaries are different, i.e. not the same life-cycle stages are included in the assessment (ICCT, 2018).

Concerning the functional unit, it is observed that the impacts should refer to a term of reference that is strictly connected to the function of the studied product. The function of xEV batteries is to accumulate energy and supply electrical current to a powertrain. According to the Product Environmental Footprint Category Rules (PEFCR) for batteries[60] (Recharge Association, 2018), the functional unit for rechargeable batteries is defined as 1 kWh of the total energy provided over the service life by the battery system. Nevertheless, this functional unit requires referring to the expectancy life of the battery, which is often hard to estimate because it is affected by many different parameters. Therefore, as underlined by Matheys et al. (2007) the choice of the functional unit (and the LCA development) is complicated when different correlated parameters have to be considered, as it is the case of traction batteries. The majority of already available LCA studies on batteries show impact results for 1 Wh of storage capacity or for 1 kg of battery, which make difficult an objective comparison between different batteries.

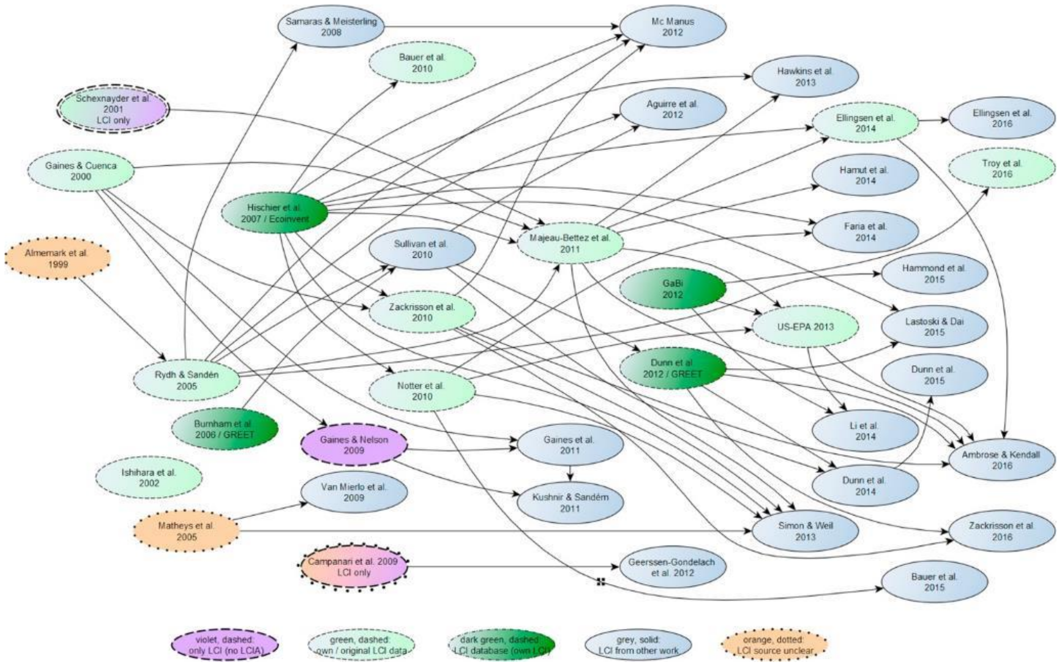
Due to the novelty of the topic, few data are available to fill in detailed and robust Life Cycle Inventories. Also, it is to be considered that the value chain of traction battery includes different

[60] http://ec.europa.eu/environment/eussd/smgp/PEFCR_OEFSR_en.htm

Countries in the world, and hence is quite complex to collect data (Lebedeva et al., 2016; Pillot, 2017). In fact, primary data (collected directly from industrial operations) could not be available for all the phases of the battery value chain. As a consequence, it could be necessary to use secondary data (from literature, database, patents, etc.). Assumptions or estimations could also be necessary to define and quantify the materials composing the battery as well as the resources spent during the production, use and end-of-life phases. Moreover, as far as the end-of-life phase is concerned, significant uncertainties are related to the current lack of consolidated solutions of reuse/recycle/disposal. Finally, important variations in the battery chemistries are expected in the next future (BOX 7).

The uncertainty of LCI is reflected on the availability and representativeness of the proxy datasets employed in the life cycle modelling for background processes. The reliability of impact results is strictly connected to the representativeness of the proxy datasets chosen for the life-cycle model. For some processes (such as, for example, the production of the water-based binder employed in EV batteries) a representative proxy is not available in life-cycle databases and assumptions have to be taken. Often, different studies refer to the same inventory and few studies adopt primary data to model the battery (Cusenza et al., 2019; Peters et al., 2017).

Figure 44. Interconnection of LCI data sources used in literature LCA studies (Peters et al., 2017)



**BOX 7: Evolution of Li-ion batteries technology**

Since 80's the electric vehicles (xEV) were on the road; due to the fast development of the technology on internal combustion engine (ICE) and the cheap, abundant gasoline, the interest on EVs steeply decrease. However, the increasing focus on the environment is one of the most important causes of a renewed attention to EVs from several perspectives (regulatory, economic, environmental, etc.).

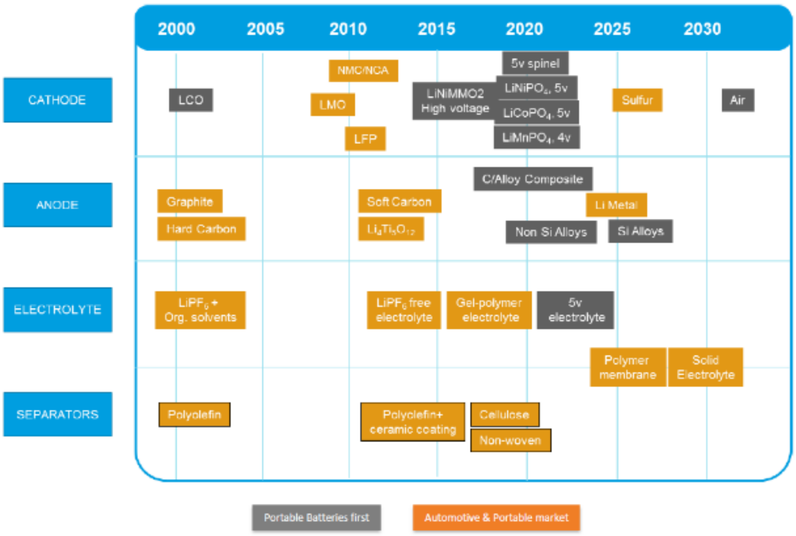
First Li-ion battery was launched by Sony in 1991 with LCO cathode. Traction batteries are evolving rapidly and new chemistries are already available in the market. Mainly driven by the batteries' performances and the cost of batteries, new materials and components are constantly under analysis.

For instance, according to recent research/communications (http://www.deq.ufv.br/arquivos_internos/eventos/NbinLiBFinalnb.tech.pdf), adding small amounts of Niobium can make Lithium Iron Phosphate (LFP) cathodes 1,000,000,000x more conductive. New Niobium materials being developed for battery anodes improve the mobility of Lithium ions by creating "spaces" in the anode material so that Lithium ions can easily move in and out of the anode, thus allowing a very high charge/discharge rate. Moreover, used with Titanium to create Titanium Niobium Oxides – TNO, new class of anode materials with approximately 3x the amount of energy storage as traditional LIBs could reduce charging times significantly.

According to (Jamesh and Prakash, 2018) also Na-ion batteries (SIBs) are becoming of interest for the next-generation power-sources, mainly because of the high abundance of Na resources that lower the cost of batteries. Another possible trend defined by Berckmans et al., 2017) is the enhancement of battery energy density through the increase of the voltage limit of cells to around 5 V, which is a harmonized voltage value used in the field of electronics. Finally, another possible trend is the reduction of safety problems through the use of solid-state electrolytes.

Long-term predictions are very difficult, but it is worth mentioning lithium-magnesium because of the superior energy density and its abundant availability, despite still being in a very early stage (Berckmans et al., 2017) (Bobba et al., 2019).

An example of roadmap of batteries for the future is provided by Pillot (2017):




Focusing on the EoL, especially due to the high impact of the manufacturing phase, EoL can play a significant role in decreasing the life-cycle impact of EV batteries, especially through recovery of materials (recycling) and/or extending the lifetime of batteries (remanufacturing/second-use). New EoL patterns are under development (section 4.1) and new recycling technologies (section 4.3.1.2), even though there is still room for improvement to better manage the waste batteries flows. The recent study of (Yun et al., 2018) identified “six research directions” concerning LIBs EoL in a life-cycle perspective:

1. Different design of batteries make complex an automatized recovery system; often the access to some components is difficult (e.g. cables or plug-in connectors of the BMS) and batteries are designed differently;
2. Battery pack disassemble requires safety awareness and specific procedures to be followed by skilled workers;
3. The recycling market need support to the adoption of oriented policies and the development of recycling technologies at industrial scale;
4. Recovery of some materials are not so economically accretive, hence the main recycling activities aim at recovering precious metals. Attention is needed for other materials that are significant in terms of environmental impact and are not commonly recovered;
5. Focusing on metallurgy processes, not all processes are already available at industrial scale due to their complexity but also the important investment required. However, their scale up could enlarge the amount and the type of materials recovered (e.g. the hydrometallurgical process allow to recover some materials that are not recoverable though pyrometallurgical processes);
6. The fast development of batteries also mean the potential adoption of different materials and components in batteries manufacturing. This aspect is a significant challenge that recyclers and recycling technology will have to face in the future.

Bearing in mind all these aspects, LCAs of vehicles, especially in case of new types of vehicles, should entail a wide set of impact categories and a sensitivity analysis focusing on the most relevant open issues. Modularity of LCAs offers to possibility of enlarging the LCA models including specific modules for new components; also, the parametrization of modules will facilitate the performance of sensitivity analyses and update of input according to the available information.

Finally, the sustainability of xEV should include technical, environmental, economic and social aspects. Focusing on traction batteries, the lack of data make this assessment quite complex to be performed. Focusing on the available information, an initial estimate of social impacts of a traction battery was performed (Eynard et al., 2018). Based on the results, and according to experts in the field (6th International Conference on Social Life Cycle Assessment, Pescara –IT, 2018), much more efforts should be applied to assess the social impacts of products in terms of both data collection and methodological assessment.

## 3.4 Lesson learnt and follow up

According to the performed analysis and the contacts with CRF group, some recommendations for future work arose.

Despite the confidentiality of information remains an issue to be faced, a stricter collaboration with industries and stakeholders of the value-chain of vehicles is strongly recommended in order to improve the data collections and, consequently, the robustness and reliability of LCIA results. The involvement of different stakeholders is quite important especially since vehicles are very complex systems.

The development of a LCA model able to assess the environmental performances of different vehicles will ease the comparison between different types of vehicles, which is currently complex due to different assumptions and system boundaries. The development of the LCA model in collaboration with an industrial stakeholder (CRF) directly involved in the life-cycle of vehicles brought the necessary knowledge to develop in detail the LCA model. Thanks to the set-up of a modular LCA, according to future changes in the automotive sector the update of the modules and/or the addition of new modules will ease also the update of the LCA model.
This is nowadays relevant since vehicles technology is changing quite rapidly also to respect the in force legislation and environmental targets. New modules can be added to the developed LCA (now focusing on conventional vehicles) to also include new components and new materials, e.g. the powertrain of xEV (electric motor and battery).

Also, the structured data collection performed by CRF and the accessibility to the IMDS permitted to establish a direct link between the data collection (inventory) and the LCA model. In addition, the adoption of parameters in all the modules of the LCA makes the model enough flexible to add and update information according to the available input data.
In case of LCA of new vehicles, a further collaboration with industrial stakeholder is important, for both the access to confidential data and also because their experience and knowledge in the field can help in identifying correspondences between IMDS and the datasets to be used for assessing the environmental impacts.

Impact assessment results highlighted that he environmental credits/debits should not be ignored in performing LCAs of vehicles. In fact, results prove the potential significance of EoL for ICEV. Considering the fast development of vehicles and the adoption of new components and materials (e.g. for xEV), the EoL contribution could be even more important.

Finally, due to the relevant contribution of passenger cars to the environmental impact of products in Europe, the extension of the lifetime of conventional vehicles in particular could correspond to significant environmental benefits. However, such potential benefits should be quantified for different impact categories; also, the changes in performance but also in technology of the replacing vehicles should be taken into account. The method illustrated in Section 2 based on the comparison of a Base-Case Scenario and a Durable Scenario represents the base for further analyses on lifetime extension of vehicles.

# Adapted Life Cycle Assessment and energy flows modelling.
# Application to the second-use of traction batteries

The relevance of taking into account the specific characteristics of products in assessing their environmental performances emerged from the scientific literature (Introduction). This is also proved by the performed assessment of the potential benefits related to extend the lifetime of products (section 2.2) but also by the modular LCAs assessing the impacts of different types of vehicles (section 3.2).

When extending the lifetime of products using energy during their use phase, the amount of energy used along the whole life-cycle should be carefully considered. In particular for these products' categories, the debate on benefits related to the lifetime extension instead of replacing used products with new and more energy performant products is still open (Introduction). Therefore, to assess the impacts in a life-cycle perspective, an in-depth knowledge of the energy flows of the system is required.

In this framework, the Pro-EnDurAncE method (chapter 2) could be applied to various products but it should be combined with the modelling of the energy flows of the system under analysis, according to the specific characteristics of products in different sectors. Based on the conclusions of chapter 3 and the fast changing of the automotive sector in Europe towards e-mobility, the extension of lifetime of batteries could be an interesting option to decrease the life-cycle impacts of traction batteries that for EVs are the most significant component for the xEV impacts.

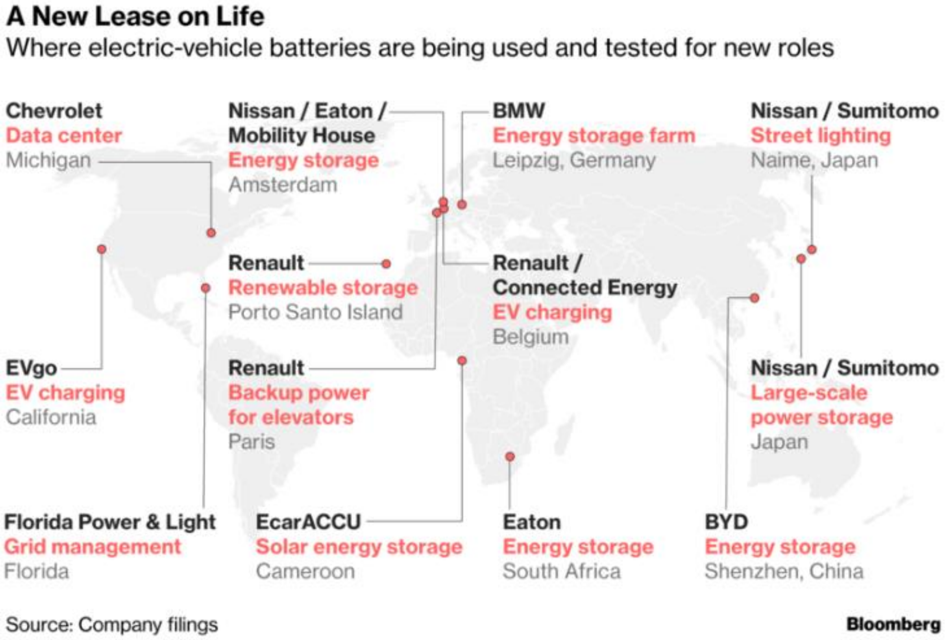
Therefore, the same approach adopted for the Pro-EnDurAncE method, i.e. the comparison between different scenarios, was adopted to develop an adapted LCA model to assess the potential environmental benefits of extending the lifetime of batteries through different strategies (Bobba et al., 2018a). To take into account the energy consumption along the lifetime of batteries, the adapted LCA includes the modelling of the energy flows of the system in which the battery is used.

In this chapter, the main outcomes of the performed literature review on extending the lifetime of batteries are illustrated (section4.1). Based on the collected information from both stakeholders of the batteries value chain and literature, the adapted LCA model was developed (section 4.2). The proposed method is then applied to traction batteries for which the lifetime is extending through its adoption in two different second-use applications (section 4.3). Main conclusions and follow up of the developed work are reported in section 4.4.

## 4.1 Literature review on second-use of traction batteries

The extension of lifetime of xEV batteries can occur through both remanufacturing and/or second-use. Currently, remanufacturing of traction batteries are not common practices in Europe as batteries are mainly addressed to recycling. Nonetheless, some companies already started to remanufacture batteries, e.g. 4R Energy Corporation in Japan[61] and some European projects are already ongoing, e.g. CarEservice[62], ABACUS[63], BatteReMan[64]. The performed mapping of existing second-use applications (Table 17) highlights the interest in the topic and show the main applications interested by potential second-use of traction batteries (the detail mapping analysis is reported in Bobba et al. (2018b)). Applications related to grid integration of renewable energy and to reserve capacity are mostly studied and they seem the most promising second-life options. Some examples of new industry of repurposed batteries in Figure 45.

Figure 45. Activities of second-use of batteries (Stringer and Ma, 2018)




Moreover, the annual number of publications focusing on EV increased since Nineties (Ramirez et al., 2018; Zhao et al., 2018), and more research are expected especially on the extension of lifetime of batteries (e.g. degradation processes, reliability of connections between EV and other energy systems (Zhao et al., 2018).

This is also confirmed by the increasing number of patents worldwide focusing on EV (Schmitt et al., 2016). The most active countries in developing EV patents are located in Asia, which also reflect the rapid increase of the EV penetration in Asian markets. Japan, China and Korea count

[61] https://www.rematec.com/news/news-articles/batteries-not-included/; https://cleantechnica.com/2018/05/15/nissan-begins-offering-remanufactured-batteries-for-leaf/

[62] http://www.careserviceproject.eu/

[63] https://blogs.warwick.ac.uk/abacus/; https://www.researchgate.net/publication/299640622_In-Service_EV_Battery_Life_Extension_Through_Feasible_Remanufacturing

[64] http://www.pem.rwth-aachen.de/cms/PEM/Forschung/Projekte/~kvia/BatteReMan/

for 50%-60% of the annual publications; note that the publications of Japan decreased in time and China publications increased. Overall, the EPO publication account for about 20% of the annual publications.
Finally, the interviewed stakeholders and the attended meeting and conferences (section 1.5.3) proved that industrial stakeholders are interested in the development of such a business and that the interest is increasing for different stakeholders of the batteries/vehicles value-chains, also including policy makers (BOX 8).

In general, second-use of batteries presents potentialities in terms of both economic[65] and environmental benefits. However, the sustainability of this option needs to be further demonstrated from different perspectives (technical, environmental, economic and social)[66]. As previously discussed for reuse wording (section 1.3), the developed work pointed out that one of the main barriers to the potential second-use of batteries is the absence of a clear and univocal definition (section 4.1.1). Moreover, other barriers have been already identified in the literature (section 4.1.2).

Concerning other aspects needed to assess the sustainability of second-use, economic information are gathered in the scientific literature. However, a detailed analysis was not performed due to the lack of data and the timing. Similarly, some initial considerations on social assessment of batteries were developed (Eynard et al., 2018), however much more efforts should be applied in this analysis (6th International Conference on Social Life Cycle Assessment, Pescara –IT, 2018). The developed work, even though more efforts are needed to obtain more robust data, show the importance of combining the social and the environmental assessments.

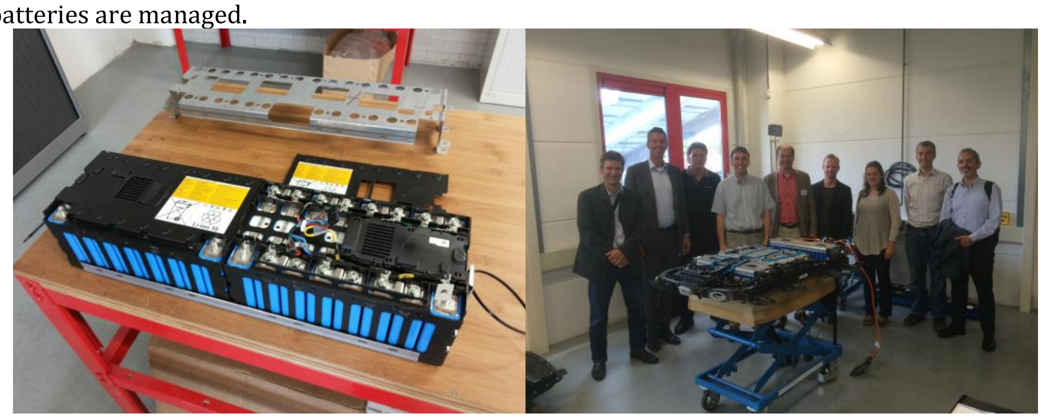
[65] E.g. the study performed by Debnath et al. (2014) estimates that 19.56% of the initial battery purchase cost can be recovered adopting repurposed batteries

[66] https://ec.europa.eu/environment/efe/content/long-term-vision-sustainable-future_en

**BOX 8: Visits on the field of industrial stakeholders working on second-use of batteries**

Autobedrijf Peter Ursem (The Netherlands) (http://www.peterursem.nl/)

Autobedrijf Peter Ursem is a car dealer who, through an environmental permit, became also a recycler and consequently a manufacturer of new products. This means that collected batteries could be tested and used for other purposes. A visit to this repurposing centre was organized and it permitted to better understand how collection, testing and repurposing of batteries are managed.



*Visit to the Peter Ursem plant (The Netherlands) (10/06/2016)*

Pampus Island (The Netherlands) (https://www.pampus.nl/en/)

Pampus Island. In the Pampus Island, one of the two batteries used for energy storage is a Li-ion battery derived from 2 xEV battery packs that were dismantled at the cell level, tested and re-assembled to be used in the island. Together with batteries, the energy requirement is covered by a PV system and a diesel generator.

*Diesel generator*

*Inverters for the PV system*

*Li-ion battery (cells and BMS)*

*Visit to the Pampus Island (The Netherlands) (06/07/2017)*

Van Peperzeel (Lelystad - The Netherlands)

Main expertise of Van Peperzeel concern the safe handling of waste batteries along the value chain (reverse logistic, sorting, and packaging for logistics). The company has developed new solutions for handling (storage/transport/packaging) Li-ion batteries especially in relation to their safety issue, solutions to prevent and extinguish fires in containers for waste batteries. They are also currently developing a new technology to fully discharge end-of-life batteries to maximize safety during recycling and recover the energy already contained in the waste battery (using the Powerwall of Tesla).

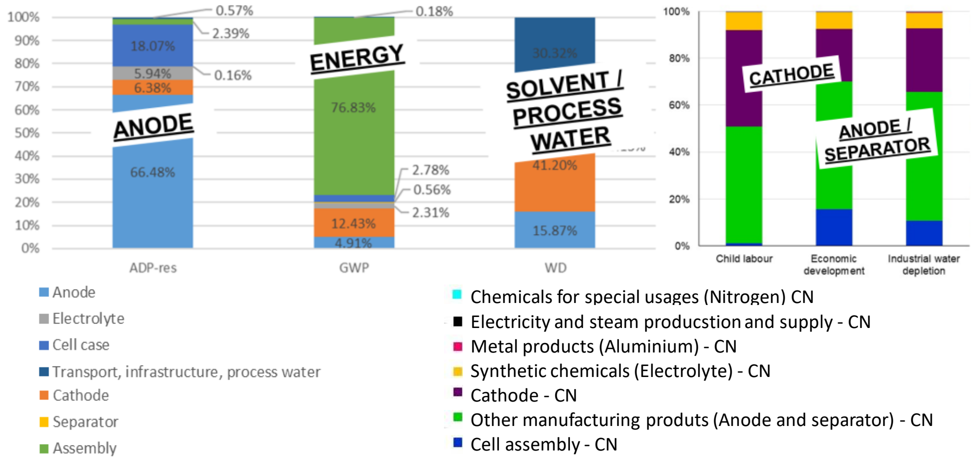
Van Peperzeel has also some manual sorting activities and then they send sorted batteries to several recyclers in Europe.

**BOX 9: Social assessment of traction batteries**

Aiming at estimate the social impacts of traction batteries, a social LCA was performed starting from the inventory used for the environmental LCA (section 3.2). The details of the study are reported in (Eynard et al., 2018), while hereinafter the most relevant conclusions of the performed analysis and the 6th International Conference on Social Life Cycle Assessment (Pescara – IT, 2018) are summarized.

The common starting point for both the analyses (social and environmental) is the LCI of materials needed in the battery cell manufacturing. However, the Bill of Materials of the product is not sufficient for modelling both the environmental and social LCAs and therefore, stakeholders should be involved to gather different types of information useful for both LCA and S-LCA. Compared to the LCA, the inventory of the S-LCA requires a broader overview of the involved processes and materials along the life-cycle and different stakeholders should be involved for the data collection, e.g. manufacturers, workers, local community.

Environmental and social LCA results – contribution analysis




Results of the performed analysis also highlighted that the geographical boundaries, often not considered as a crucial aspect for LCA, should be considered when assessing social aspects in specific Countries involved in the supply chain. It is therefore possible to identify the most critical sites along the supply chain from both the social and the environmental perspectives. For that, site-specific data collected from the supply chain are needed in order to minimize the uncertainty related to the generic data provided by databases.

Also, the combination of both environmental and social LCIA in case of emerging technology such as Li-ion batteries could offer a wider overview of impacts of products for which strategic materials for Europe are used (e.g. CRMs). End-of-Life processes for some relevant materials for the market (e.g. recycling of aluminium, nickel, cobalt) can mitigate the environmental burdens related to these materials but also cause positive/negative social impacts in specific areas (e.g. job creation, illegal shipment to third Countries from Europe). Then, a further development of the study should include the End-of-Life of cells.

Table 17: Recent activities and studies using second-life xEV LIBs for several second use applications

| Paper<br>Actors [Reference] | Second Use Applications | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| | **Transmission & Distribution upgrade deferral** | **Energy arbitrage / energy time-shift** | **Area regulation / regulation service / frequency regulation)** | **Reserve capacity / Supplemental reserve / Backup supply** | **Grid System** | **Load levelling / Load shifting** | **Peak shaving/smoothing** | **Power quality** | **Demand charge management** | **Renewable Energy Sources (RES) Integration (e.g. PV Firming, power smoothing)** | **EV charging** |
| **Peer-reviewed scientific publications and other studies** | | | | | | | | | | | |
| (ADEME, 2011) | | | | X | | | | | | X | |
| (Neubauer and Pesaran, 2011) | | | X | | X | | | X | | | |
| (Viswanathan and Kintner-Meyer, 2011) | | | X | | | | | | | | |
| (Tong et al., 2013) | | | | | | | | | | X | |
| (Ahmadi et al., 2014b) | | | | | | X | | | | | |
| (Tamiang and Angka, 2014) | | | | X | | | | | | X | |
| (Faria et al., 2014) | | | | | | X | X | | | | |
| (Heymans et al., 2014) | | | | | | X | | | | | |
| (Koch-Ciobotaru et al., 2015) | | | | | | | | | | X | |
| (Canals Casals et al., 2015) | | X | | | | | | | | X | |
| (Neubauer et al., 2015b) | | | | | | | X | | | | |
| (Richa et al., 2015) | | | | X | | | | | | | |
| (Saez-de-Ibarra et al., 2015) | | | | X | | | | | | | |
| (Sathre et al., 2015) | | | | | | | | | | X | |
| (Cready et al., 2003) | | | | X | X | X | | | | | |
| (Narula et al., 2011) | X | X | X | | | | | | | | |
| (Neubauer and Pesaran, 2011) | X | | X | | | | | X | | | |
| (Williams and Lipman, 2011) | X | X | X | X | | X | | | X | X | |

| **Industrial activities** | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **European (EU)** | | | | | | | | | | | |
| Daimler, The Mobility House, GETEC, REMONDIS (Morris, 2015a) | | | | | X | | | | | X | |
| Bosch, BMW, Vattenfall (Bosch, 2016; Kane, 2016) | | | X | | X | | | | | | |
| Nissan and Eaton (EATON.; Morris, 2016a) | | X | | X | | | | | | | |
| Renault and Connected Energy (Morris, 2016b) | | X | | | | | | | | X | |
| EDF, Forsee Power, Mitsubishi Motors Corp., Mitsubishi Corp. (Forsee Power, 2015) | | | | | | | | | | X | |
| **International** | | | | | | | | | | | |
| GM and ABB, and Nissan with Sumitomo/ABB (Williams, 2011) | | | | X | | | | | | | |
| 4R Energy (joint venture between Nissan and Sumitomo Corporation) (Gordon-Bloomfield, 2015) (Sumitomo, 2014) | | X | | | | | X | | | | |
| BMW and BECK Automation (Morris, 2016c) | | | | | | | | | | X | |
| FreeWire Technologies and Siemens (Morris, 2015b) | | | | | | | | | | | X |
| Spiers New Technologies (Ruoff, 2016; Technologies, 2015) | | | | X | | | X | | | | |

**R&D activities**

| | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|
| **EU-funded projects** | | | | | | | | | | | |
| ABattReLife (ABattReLife.) | No specific application was defined | | | | | | | | | | |
| AlpStore (Alpstore.) | | | | | X | | | | | | |
| Batteries2020 (Batteries2020.) | | | | | | | | | X | X | |
| Energy Local Storage Advanced system (ELSA) (ELSA, 2017) | X | X | X | X | X | X | X | X | X | X | |
| Netfficient (NETfficient - Storage for Life.) | No specific application is currently defined (at the moment this report was written) | | | | | | | | | | |
| 2Bcycled (ARN, 2014) | | | | | | | | | | X | |
| **International** | | | | | | | | | | | |
| Batteries Second Use (B2U) – NREL (Center for Sustainable Energy, 2016; NREL (National Renewable Energy Laboratory), 2015) | | | X | | | | | | X | X | |
| | 4 | 7 | 8 | 10 | 6 | 6 | 5 | 3 | 4 | 15 | 1 |

### 4.1.1 Definition of second-use for batteries

A general issue underlined by the interviewed stakeholder is the absence of a clear definition of “second life application”. A standardised and recognised definition of “second life application” within the regulatory framework could support the future strategies in extending batteries’ lifetime and creating new investments opportunities.

Moreover, imprecise or interchangeable terminology are often in various documents (Hartwell and Marco, 2016; James Paul et al., 2015; Recharge, 2014). Based on both the BSI British Standards Design for manufacture, assembly, disassembly and end-of-life processing (BS 8887-2:2009), and on DIN EN standards, the APRA Europe organization (Automotive Parts Remanufacturers Association) defined and distinguished between different terms used in the framework of remanufacturing (APRA Europe, 2012). As schematized in Figure 46, repurpose is defined as “to use a product for a different purpose than originally intended. An item can be repurposed by modifying it to fit a new use, or by using the item as it is in a new way”. This is also aligned to Ardente et al. (2018), who state that ““reuse” implies that a product is being utilized for the purpose for which it was conceived, and “repurposing” refers to utilizing products in other, different applications (often referred to as “second-use” applications)”.

James Paul et al. (2015) define the battery repurposing as a process involving “the breakdown of packs into modules, inspecting the hardware of the modules, performing inspection and health benchmark tests on the modules, and certifying that the modules meet a market-defined second-life standard. Once the modules have been certified, the second process, repackaging, takes place. The repackaging process involves putting modules deemed “good enough” for second-use into sub-packs and packs that can be shipped for use in stationary systems”. In this process, it is possible that very good modules can be used again for EVs (Ruoff, 2016). Note that the analysis performed by Neubauer et al. (2015a) identified the technician labour as the major cost element of repurposing.

Hartwell and Marco (2016) discussed the ambiguity deriving by the absence of an exact meaning of “related circular economy activities” among which refurbishment and remanufacturing are included. ‘Warranty’ and ‘design-life’ were identified as concepts able to provide a clear definition of remanufacturing and, consequently, to propose definitions also for refurbishment of battery packs.

RECHARGE, the European Association for Advanced Rechargeable Batteries[67], aiming at defining ’re-use and second-use’ of batteries, proposed to establish a set of minimum requirements that need to be fulfilled before authorising the re-use or the second-use of batteries after a first service life. A non-exhaustive list of minimum requirements, as shown in Table 18, shall be met in order for RECHARGE to facilitate the re-use. RECHARGE only supports the second-use of batteries when the battery remains under the responsibility of the producer acting as the first entity placing the battery on the market. In absence of a legal basis and clear minimum requirements, second-use is not

[67] www.rechargebatteries.org

supported by RECHARGE, as there are too many unknown factors that could impact the reliability of the product and safety of the end user (Recharge, 2014).

Figure 46: The potential life-cycle(s) of a product and its materials (The likely change in quality level compared to the original product is given in parentheses) (APRA Europe, 2012)

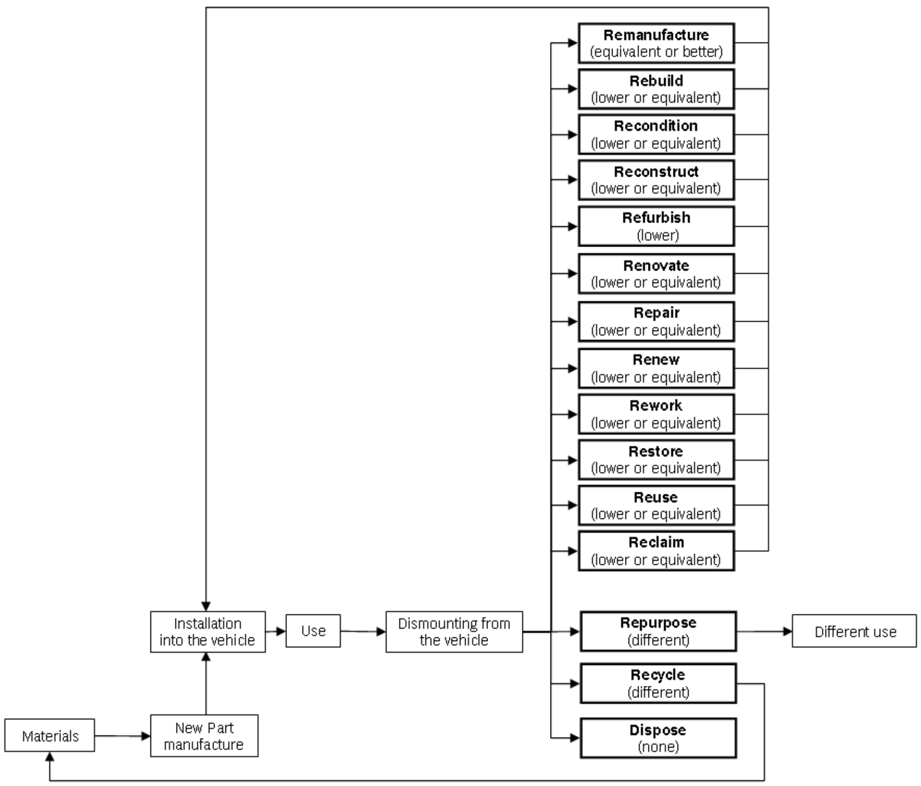



Table 18 Indicative list of minimum requirements to be considered for allowing re-use or second-use of batteries (adapted from (Recharge, 2014))

| Proposed Minimum Requirements for | |
|---|---|
| **Re-use (identical use)** | **Second-use** |
| **Application**<br>- Re-furbishment or re-conditioning by qualified professional<br>- Control of equivalent performances, e.g. through the BMS<br>- Quality, Safety and Performance standards to be observed<br>- Etc… | In absence of a legal basis, additional criteria might be required – e.g.<br>- Compatibility issue between 1st and 2nd application |
| **Producer Responsibility**<br>- Producer identified<br>- Warranty offered by producer | - Responsibility for the technical performances<br>- Producer responsibility to be defined: technical and EoL |
| **Safety**<br>- Technical requirements maintained<br>- Safety standards respected (tests) | - Compliance with safety testing requirements before second-use |

### 4.1.2 Barriers to second-use of batteries

However, this EoL option is challenged by the existence of some barriers, e.g. regulatory/economic/technical barriers, safety and responsibility issues. In this context, more efforts are required to provide “an adequate legal framework for second-life applications”, for example in the forthcoming review of the batteries Directive (EC, 2017c).

Contacts with stakeholders revealed that one of the most relevant barriers to be faced is represented by the absence of a clear definition of “second-use application” and of a legal framework supporting this option, as above mentioned. Furthermore, Table 19 gives an example of important factors influencing the potential reuse on xEV batteries in second-use applications. a more detailed description of the identified barriers is provided by Bobba et al. (2018b).

Table 19: Indicative list of factors influencing the potential reuse on xEV batteries in second-use applications as identified in a few reports (non exhaustive).

| Reference | Regulatory barriers | Technical barriers | Safety issues | Economic barriers | Responsibility issues |
|---|---|---|---|---|---|
| (Deloitte, 2015) | X | | | | |
| (Kempener and Borden, 2015) | X | X (Performance issues) | X | X (Lack of monetary compensations schemes available for the benefits of battery storage system) | |
| (Elkind, 2014) | X (Complex and adverse regulatory structures that limit market opportunities and increase costs (difficulties in transporting batteries as classified as hazardous waste; existence of incentives that indirectly discourage the second-use of batteries; uncertainty about safety issues of second-use of batteries)) | X (Lack of data about battery performance in both first and second life applications) | | X (Uncertain economic return and market for many energy storage applications<br>Potential future competition between repurposed batteries applications and new energy storage technologies<br>Potentially expensive repurposing or redesigning of the battery pack for new applications<br>High repurposing costs may limit opportunities for financing. Economic uncertainty about second-life battery value translating to reduced upfront costs for electric vehicle consumers) | X (Liability concerns about which entity is responsible for second-life batteries once they complete their first life in the vehicle) |
| (Neubauer et al., 2015a) | X (Utilities and regulators should develop policies that encourage the use of ESS) | | | X (No economic incentive to replace a PEV battery prior to the end of the original vehicle’s service life (approximately 15 years)<br>Technician labour is a major cost element of repurposing operations that must be minimized) | |
| (Richa et al., 2015) | | | | X (Second-life batteries are currently ineligible for incentive programs or federal investment tax credits for | |

| Reference | Regulatory barriers | Technical barriers | Safety issues | Economic barriers | Responsibility issues |
|---|---|---|---|---|---|
| | | | | grid storage, onsite, or residential energy storage systems in the USA) | |
| (Canals Casals et al., 2015) | | | | X (The best possibility to reach a positive economic balance is the direct re-use of the batteries without module manipulation) | |
| (Ahmadi et al., 2014b) | | X (Difficulties and uncertainties in establishing specific parameters for the analysis (e.g. lifetime, capacity of batteries in the future, driving patterns, etc.) Customers attitudes affect some technical aspects of xEV batteries (driving attitude, perception of costs, batteries retirement, etc.)) | X (Battery removal poses hazards associated with high voltage safety and handling of liquid coolant) | | |
| (Reinhardt et al., 2017) | X (Unclear and undefined legislation) | X (High volumes of waste xEV batteries) | | X (Profitability of recycling processes) | |

Note: "X" means that the barriers / issues are found relevant in the study. More explanations, when relevant, are given between brackets.

## 4.2 Environmental assessment of extending the lifetime of traction batteries

Despite various papers focus on the environmental impact of second-use applications of xEV batteries[68], guidelines or harmonized approaches do not exist yet and the comparison between the LCA results are often complicated. In fact, major differences are observed between the studies, especially concerning differences in the assessed applications, different life-cycle stages included in

[68] Examples of applications assessed in the literature are: smoothing for renewable energy systems, energy storage of a single wind turbine/photovoltaic/battery system, off-grid photovoltaic vehicle charging system; diurnal energy shifting, allowing expanded use of intermittent renewable energy sources such as wind and solar, load shifting and peak shaving.

the assessment, lack of inventory data to model the impacts of the life-cycle stages and the impact methods used to assess the impacts of the system.

The adoption of batteries in combination with renewable energy installation in buildings sounds the most promising application (Table 17). Also, note that the adoption of a repurposed EV battery in storage applications could substitute batteries based on non Li-ion chemistries (e.g. lead-acid batteries) or different sources of energy (e.g. consumption of fossil fuels) and support a shift to renewable sources of energy (Canals Casals et al., 2015; Neubauer et al., 2015a; Richa et al., 2015; Sathre et al., 2015).

The use stage of batteries is recognised as an important stage to be assessed (Canals Casals et al., 2015; Richa et al., 2015). The performance of the battery in a specific system depends on both the batteries characteristics (e.g. battery chemistry, capacity, efficiency) and the system in which they are adopted (grid-connected, stand-alone, power/energy application) (Koch-Ciobotaru et al., 2015; Weniger et al., 2014b). Due to the absence of primary data, in place of data reflecting real energy systems, often average data, estimations and assumptions are used (Ahmadi et al., 2014b; Richa et al., 2015). Energy flow of the system should be assessed through the daily production and demand curves of the renewable system (Weniger et al., 2014b), also due to the significant effects of user behaviour in changing the proportion of renewable household energy and, therefore, on the energy flows of the system (Hinterstocker et al., 2017).

In LCA, the system boundaries characterizing the study should be clearly defined (ISO, 2006a). Concerning second-use of batteries, different approaches can be observed in the literature. For instance, aiming at assessing the whole life-cycle of the xEV battery, all the life-cycle stages on the xEV battery, i.e. car manufacturing, use of the battery in both the car and in the second-use application, the battery recycling (Canals Casals et al., 2015; Richa et al., 2015). Other authors consider only the life-cycle stages directly affecting the second-use of the xEV batteries (Faria et al., 2014; Sathre et al., 2015). Also, due to limited data available, the repurposing stage is often modelled through assumptions considered as negligible from an environmental perspective (e.g. in Canals Casals et al. (2015) and Faria et al. (2014)). Even though battery testing is expensive and time consuming (Nenadic et al., 2014; Neubauer et al., 2015a), a detailed understanding of the battery behaviour is needed (DeRousseau et al., 2017; Koch-Ciobotaru et al., 2015).
Also related to the system boundaries of the study, it is worthy that the energy mix used in the assessment could heavily affect the LCIA results, as well as the meteorological conditions (DeRousseau et al., 2017; Erkisi-Arici et al., 2017; Faria et al., 2014).

### 4.2.1 Method for the environmental assessment second-use of EV batteries

Similarly to the environmental assessment of durability of products, also in this case the method is based on the comparison of the environmental impacts of two different scenarios from a life cycle perspective; the impacts of the scenarios are assessed based on the LCA (section 4.2). The life cycle stages of the scenarios defined in this method are depicted in Figure 47.

The “Repurposed Scenario” (Figure 47a) assumes that, after its first applications in an EV, the EV battery is reused in a second application. The environmental impact of the Repurposed Scenario refers to all the life-cycle stages involved in the second use of the EV battery, i.e. battery manufacturing ($P_{EVB}$), battery repurposing ($Rep_{EVB}$), battery use in the storage application ($U'_{EVB-stor}$) and battery EoL ($E_{EVB}$).

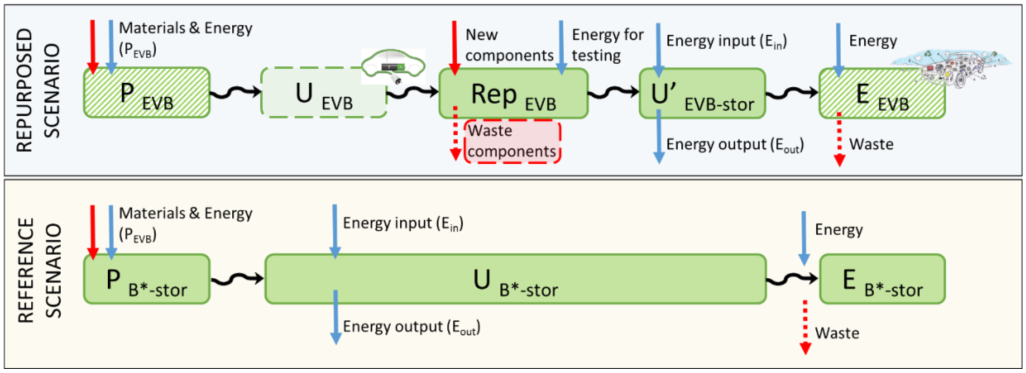
The use of the battery in the EV affects the battery characteristics and its lifetime in the second-use application (e.g. residual capacity, battery efficiency after the use in the EV); the impact of this stage ($U_{EVB}$) is not included in the assessment since this is not directly related to the second-use application (dashed box in Figure 47). Nevertheless, the first use of the battery affects its second-use, especially in terms of performances and lifetime. During the repurposing stage some components of the battery pack can be replaced (e.g. casing) by new ones. In this case, the impact of the waste components (dashed box in Figure 47) is not included in the assessment of the Repurposed Scenario, since these waste are assumed to relate exclusively to the first application of the EV battery, and therefore these are out of the system boundaries of the analysis of the repurposed battery. Consistently, the impacts of the manufacturing and the EoL of the new components used during the repurposing are fully allocated to the second use of the EV battery.

Concerning the manufacturing and the EoL of the battery in the Repurposed Scenario (striped boxes in Figure 47), the environmental impacts of these two stages ($P_{EVB}$ and $E_{EVB}$) should be allocated between the different applications along the whole life cycle since they refer to both the first application in the EV and second application in the storage system. Therefore, not all the impact of these two stages should be fully allocated to the second use of the EV battery.

The Repurposed Scenario is compared to a “Reference Scenario” (Fig. 1b) in which a fresh battery is used for storing energy flows. The impacts of the Reference Scenario consist of the impacts of the battery manufacturing ($P_{B*-stor}$), the use of the battery in the system ($U_{B*-stor}$) and the battery EoL ($E_{B*-stor}$). Note that the method can be adapted to consider different configurations.

The adoption of different batteries (e.g. in terms of capacity, chemistry, etc.) in a system will affect the overall energy flows of the system. Since the aim of the analysis is to assess the potential environmental benefits of using a repurposed battery in a specific system, in both Scenarios the impacts related to the use of the battery in the system ($U'_{EVB-stor}$ and $U_{B*-stor}$) refer to the impacts of all the input and output energy flows ($E_{in}$ and $E_{out}$) of the system along the lifetime of the battery. This aspect is highly dependent on the characteristics of the system, including geographical (e.g. local grid mix, temperature) and technical considerations (e.g. residual capacity of the EV battery, drivers behaviour, load profile of the building).

Figure 47: Schematic presentation of the two scenarios to be compared for the assessment of EV repurposed batteries in a life cycle perspective. Dashed boxes represent stages/processes not included in the analysis, whereas striped boxes represent stages partially included in the analysis.




Note that $U'_{EVB-stor}$ identifies the second-use of the EV battery after its repurposing. "B*" identifies a battery not specifically realized to be used in EV but usable in storage applications.

Lifetime in the stationary application of fresh batteries is usually longer than lifetime of repurposed batteries. However, the impacts of different Scenarios have to be compared only through a consistent functional unit (ISO, 2006a). Lifetime of batteries, and consequently energy flows of the assessed system, depends on both batteries' characteristics and applications. Therefore, the functional unit of both Scenarios is represented by the average yearly energy balance of the system in which the battery stores energy and this is used for their comparison. For such a purpose, the life-cycle impacts of both the Repurposed and the Reference Scenario are divided by the lifetime of the battery in the assessed application.

#### 4.2.1.1 Repurposed Scenario

In the Repurposed Scenario, the allocation of the environmental impacts of the manufacturing ($P_{EVB}$) and the EoL ($E_{EVB}$) of the repurposed EV battery along the whole life cycle are modelled through the adoption of two allocation factors ('α' and 'β'). The average yearly impacts of the Repurposed Scenario ($I_{Reuse\ Scenario}$) are calculated as follow:

$$I_{Repurposed\ Scenario,n} = \frac{\alpha \cdot P_{EVB,n} + Rep_{EVB,n} + U'_{EVB-stor,n} + \beta \cdot E_{EVB,n} + E_{EVB\ new\ components,n}}{T_{EVB-stor}} \qquad (1)$$

Where:

- $I_{Repurposed\ Scenario,n}$ = impact of category "n" for the Repurposed Scenario [unit/time];
- $P_{EVB,n}$ = impact of category "n" for the EV battery manufacturing [unit];
- $Rep_{EVB,n}$ = impact of category "n" for the EV battery repurposing [unit];
- $U'_{EVB-stor,n}$ = environmental impact of category "n" for the energy use in the storage system in which the EV battery is used [unit];
- α = allocation factor considering the impact of the EV battery manufacturing to be allocated to the second use [-];
- β = allocation factor considering the impact of the EV battery EoL to be allocated to the second use [-];

- $E_{EVB,n}$ = impact of category "n" for the EV battery EoL [unit];
- $E_{EVB\ new\ components,n}$ = impact of category "n" for the EoL of the new EV battery components [unit];
- $T_{EVB-stor}$ = lifetime of the EV repurposed battery storing energy in the storage system [time].

The impact of the repurposing stage consists of the impacts of the different operations, as the transports related to the EV battery collection ($TR_{B-car}$), the testing of the EV battery implying some energy consumption ($U_{testing}$) and the EV battery checking including the possible substitution of some components ($P_{new\ components}$). The impact of the EV battery repurposing is calculated as:

$$Rep_{B-car,n} = TR_{B-car,n} + U_{testing,n} + P_{new\ components,n} \quad (2)$$

Where:

- $TR_{EVB,n}$ = impact of category "n" for the EV battery collection [unit];
- $U_{EVB\ testing,n}$ = impact of category "n" for the EV battery testing [unit];
- $P_{EVB\ new\ components,n}$ = impact of category "n" for the replacement of components of the EV battery [unit].

#### 4.2.1.2 Reference Scenario

The average yearly impacts of the Reference Scenario ($I_{Reference\ Scenario}$) are calculated as:

$$I_{Reference\ Scenario,n} = \frac{P_{B^*-stor,n}+U_{B^*-stor,n}+E_{B^*-stor,n}}{T_{B^*-stor}} \quad (3)$$

Where:

- $I_{Reference\ Scenario,n}$ = impact of category "n" for the Reference Scenario [unit/time];
- $P_{B^*-stor,n}$ = impact of category "n" for the battery manufacturing [unit];
- $U_{B^*-stor,n}$ = impact of category "n" for the energy use in the storage system in which the battery is used [unit];
- $E_{B^*-stor,n}$ = impact of category "n" for the battery EoL [unit];
- $T_{B^*-stor}$ = lifetime of the battery storing energy in the storage system [time].

**<u>Impacts of the use of the battery</u>**

Consistent with the main goal of the proposed method, the impact of the use of the battery in a storage application is assessed through the assessment of the input/output energy flows of the system. For both the Repurposed and the Reference Scenarios the impacts of the use stages ($U'_{EVB-stor}$ and $U_{B*-stor}$) are different according to the battery characteristics and the configuration of the system in which batteries are used. Differences refer, for instance, to energy losses related to the battery, energy requirements of the system, energy exchanges with the grid. This requires the assessment of the energy flows in order to evaluate the overall input and output flows ($E_{in}$ and $E_{out}$) of the specific system.

The environmental impacts of the use stages of the two Scenarios are calculated as the difference between the impacts of these flows:

$$U_n = (E_{in} - E_{out}) \cdot u_n \quad (4)$$

Where:

- $E_{in}$ = energy entering the system (e.g. from the grid) [kWh];
- $E_{out}$ = energy leaving the system (e.g. to the grid) [kWh];
- $u_n$ = environmental impact of category "n" per kWh of energy [unit/kWh].

According to the specific characteristics of the system, Formula (4) refers to both the Repurposed ($\mathrm{U_n} = \mathrm{U'}_{\mathrm{EVB-stor}}$) and Reference Scenario ($\mathrm{U_n} = \mathrm{U}_{B^*\mathrm{-stor}}$).

#### 4.2.1.3 Accounting for the environmental benefits

The benefits/drawbacks of the adoption of a repurposed EV battery in a specific application are assessed through the difference between the life-cycle impacts of the Repurposed and the Reference Scenarios:

$$\Delta_{reuse,n} = I_{Reference\ Scenario,n} - I_{Repurposed\ Scenario,n}$$

Environmental benefits in substituting a fresh battery with a repurposed battery occur when $\Delta_{reuse} > 0$, i.e. $I_{Reference\ Scenario,n} > I_{Repurposed\ Scenario,n}$.

Finally, in order to ease the interpretation of results and to assess the relevance of the impacts in the different scenarios, an index is introduced as the ratio between the $\Delta_{reuse}$ and the impacts of the Reference Scenario.

$$D_{reuse,n} = \frac{\Delta_{reuse,n}}{I_{Reference\ Scenario,n}} \cdot 100 \quad [\%]$$

For example, a value of $D_{reuse,GWP}$ of 10% means that reusing the EV battery in the energy storage systems would allow a reduction of 10% of the life cycle GWP compared to the Reference Scenario.

#### 4.2.1.4 Modelling aspects and relevant factors

Thanks to the flexibility of the developed model different aspects affecting the environmental impact of both the Reference and the Repurposed Scenarios can be considered in the modelling, for instance allocating the impacts of the manufacturing and EoL of the battery to its second use, and assessing different configurations of the system. Due to complexity of these aspects, some simplifications options are hereinafter illustrated.

**<u>Allocation rules</u>**

Energy and environmental assessment of reusing products implies that the impacts of some life-cycle stages (e.g. production and EoL) affect both the first and the second application of products.

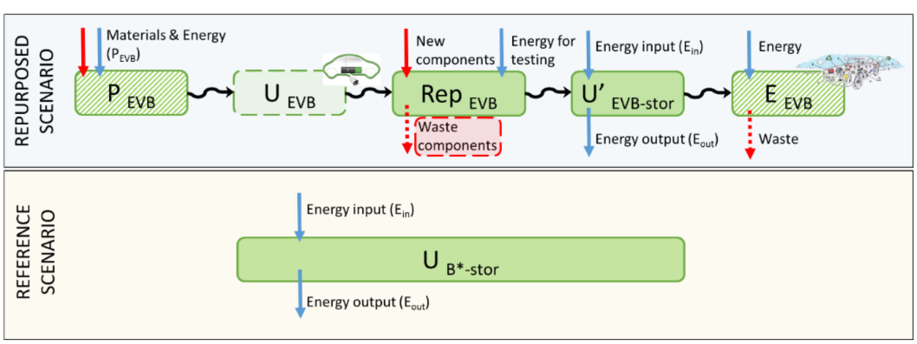
Impacts allocation is used for solving this issue (ISO, 2006a; ISO 14044:2006, 2006). In general, different criteria could be adopted to determine these two coefficients, including physical parameters (e.g. energy content, mass) or economic considerations (e.g. market price) (Wolf et al., 2012). Available allocation solutions for modelling the environmental performances of the EoL stage of products have been discussed by Allacker et al. (2014) in order to assess their suitability in the framework of EU products policies. Even though reuse is recognised as relevant for all the assessed methods, how to address the environmental modelling of reuse and how to solve multi-functionality in LCA is not still clearly defined and currently depends on the allocation decisions of the LCA practitioners (Allacker et al., 2014; Pelletier et al., 2015; Richa et al., 2015).

According to the current European legislation, after their first use in EV, batteries are classified as "waste", i.e. there is not yet a developed market for the reuse of EV batteries for second-use applications in Europe. Then, according to AFNOR - *Association française de normalisation* cited in Allacker et al. (2014), "*if the raw materials market is in disequilibrium because producers are demanding secondary raw materials which are in short supply, then there are grounds for offering incentives to producers of recycled products in order to pull the market. All of the EoL impacts are allocated to the producer*". In this case, the environmental impact of EV battery manufacturing and EoL should be fully allocated to the first life (i.e. $\alpha = \beta = 0$). However, with the potential future development of a business case as stated by some authors (Ahmadi et al., 2014b; Neubauer et al., 2015a; Ruiz et al., 2016), the battery could be manufactured focusing also on its potential second-use application; so that, 'α' and 'β' coefficients could potentially be not null in the future when a market will be established for re-purposed batteries.

**<u>Different system configurations</u>**

Finally, it is observed that the Scenarios could be defined accordingly to the goal of the assessment. As an example, the repurposed EV battery could be adopted in a system where no batteries are used. In this case, the impacts of the Reference Scenario ($I_{\text{Reference Scenario}}$) do not include the impacts related to both the manufacturing and the EoL of the battery (i.e. $P_{B^*-\text{stor}} = E_{B^*-\text{stor}} = 0$), and it will be equal to the impact of the energy use in the system ($U_{B^*-\text{stor,n}}$) (Figure 48). The EV repurposed battery could also be adopted in a system that is not connected to the grid (e.g. stand-alone building). In this case, the environmental impact per kWh of energy ($u_n$) refers to an energy source that is different from the grid mix (e.g. diesel, natural gas).

Figure 48: Schematic presentation of the two Scenarios in case of a stand-alone building without any batteries



# 4.3 Environmental assessment of second-use of a LMO/NMC battery

Based on the literature review (section 4.1) and the availability of primary data, two different applications in which a repurposed LMO/NMC battery is adopted, i.e. increase of PV self-consumption (section 4.3.1) and peak shaving (section 4.3.3).

For both case-studies, the impacts of the LMO/NMC battery as illustrated in section 4.3.1.3 are reported. The most relevant characteristics for sizing the configuration assessed in the environmental assessment are summarized in Table 20.

Table 20: Battery characteristics

| Parameter | LMO /NMC Repurposed battery | LMO /NMC Fresh battery | Source of the information |
|---|---|---|---|
| Chemistry | LMO/NMC: 0.52 $LiMn_2O_4$ + 0.48 $LiNi_{0.4}Mn_{0.4}Co_{0.2}O_2$ | | Derived from lab tests |
| Nominal capacity of the battery [kWh] | 11.40 (300V - 38Ah) | | Manufacturer |
| Number of cells per modules / per battery | 8 cells/module; 80 cells/battery | | Manufacturer |
| Initial RTE (Round-trip efficiency)[69] [%]+ | 98% | >98% | Based on (Görtz, 2015) and own measurement (section 4.3.1.2) |
| Initial capacity for the assessment [%] | 81.31% | 100% | Derived from lab tests |
| End-of-second-use Retained Capacity [%] | 60% | | Based on (Canals Casals et al., 2015; Lacey et al., 2013; Oliveira, 2017) |
| Battery degradation | -3 Wh/cycle (cycling aging); -0.13 Wh/day (calendar aging) | | Based on (Faria et al., 2014) Derived from lab tests |

\+ a linear decrease of the battery efficiency is considered (5 percentage points in 5 years)

Since the impact of the use phase is strictly depending of both the battery and the application characteristics, the energy flows of both applications are studied in detail and then used as input for the analysis. Sections 4.3.1 and 4.3.3 give a short overview of the selected applications and of the data used as input for the use phase impact assessment.

## 4.3.1 LCA of an EV battery pack

According to the main outcomes of the performed literature review, primary data on traction batteries are lacking and several authors based their inventories and assumptions on previous studies (Thomas et al., 2018). Moreover, differences in assumptions for various LCA studies make the comparison between results complex (section 4.1).

[69] RTE is represents the total energy output (at discharge) divided by the total energy input (at charge) measured between the same state-of-charge (SoC) end points associated with the application of the duty cycle during the test. It is expected that it may fade during the life test

In the framework of the SASLAB project, primary data were derived from the dismantling of a battery pack and then complemented by literature. In the following, the LCA of the battery pack is reported. For more details, see (Bobba et al., 2018b).

#### 4.3.1.1 Goal and scope

The aim of this LCA is to assess the environmental performances of a battery pack available in the market. This LCA will be then used to perform an environmental assessment of the adoption of xEV batteries in second-use applications. The performed LCA is compliant to the international standards (ISO, 2006a; ISO 14044:2006, 2006), and is used to publish a dataset compliant with the ILCD entry level requirements[70].

The analysed product is the Mitsubishi Outlander Plug-In Hybrid Electric Vehicle (PHEV) battery pack (Figure 49). It weighs 175 kg and consists of 10 modules, each made up of 8 battery cells. Each cell has a nominal voltage of 3.75 V and a capacity of 38 Ah. The 80 cells are connected in series providing a nominal voltage of the battery pack of 300 V and a total nominal capacity of 11.4 kWh. The cell has a cathode based on 0.52 $LiMn_2O_4$ + 0.48 $LiNi_{0.4}Mn_{0.4}Co_{0.2}O_2$ (LMO/NMC lithium - ion battery)[71] and an anode based on graphite. The functional unit (FU) of the study is an LMO/NMC Lithium-ion battery pack for PHEVs.

The LCA of the case-study product is performed through SimaPro 8.3 software and the database used is Ecoinvent 3. All material components are modelled as 100% of primary production.

The recommended ILCD/PEF recommendations (EC - JRC, 2012) are used for the LCIA. Note that, according to previous JRC studies, the land use, the water resource depletion and ionizing radiation impact categories have been excluded due to limited life-cycle inventory data[72] (Bobba et al., 2015; Latunussa et al., 2016) and the Resource Depletion impacts have been specified into the Abiotic Depletion Potential, mineral resource impact category[73] (Bobba et al., 2015). Finally, Cumulative Energy Demand method (Frischknecht et al., 2007) is also included in the assessment.

[70] http://eplca.jrc.ec.europa.eu/LCDN/

[71] Coefficients refer to the weight fraction

[72] According to the ILCD guidelines the ionizing radiation is classified as "interim" (best among the analysed methods for the impact category, but still not ready to be recommended); land use and water resource depletion are classified as "level III" (recommended, but to be applied with caution)

[73] The abiotic depletion potential - resources - is an impact category that account for the extraction rate of a certain resource (in relationship to the estimated world reserves), compared to a reference resource (antimony).

Figure 49: Mitsubishi Outlander PHEV battery pack (visit to the Peter Ursem plant, The Netherlands)

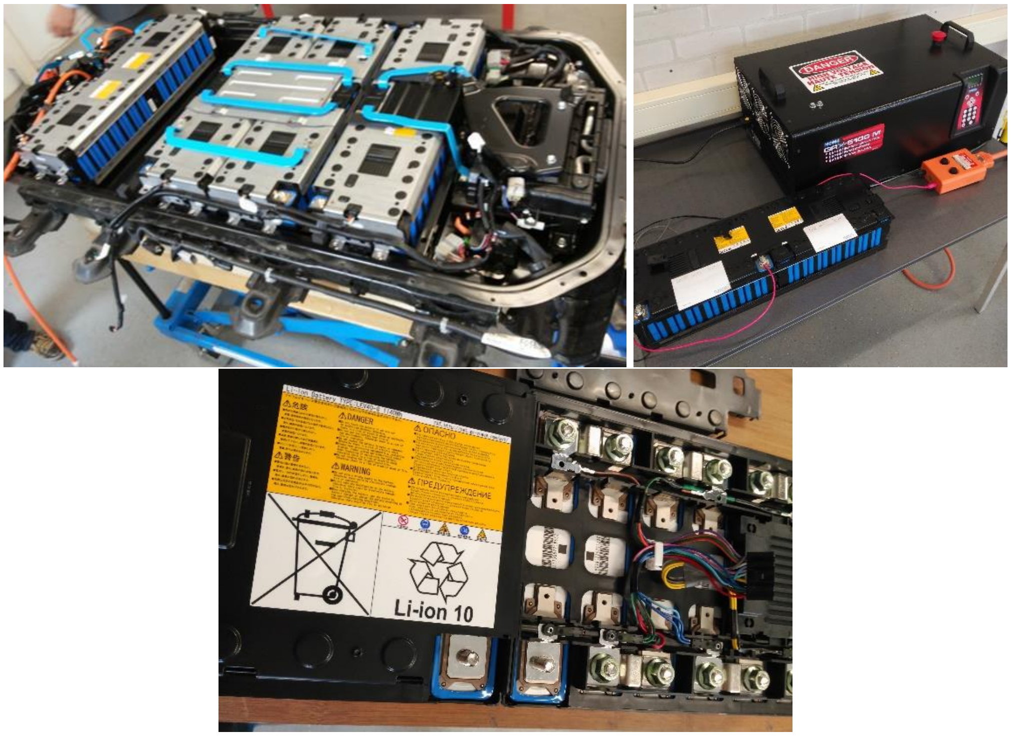



Figure 50: Dismantling of the Mitsubishi Outlander PHEV battery pack

### 4.3.1.2 Life Cycle Inventory (LCI)

**Manufacturing**

To model the manufacturing step, the battery components have been clustered in four main groups: battery cells, battery packaging, battery management system (BMS), and cooling system (Figure 51).

Colleagues in JRC Petten provided the BoM through the cells dismantling, weighting and classification of materials. Detailed LCI is provided in Bobba et al. (2018b) and Cusenza et al. (2019). The upstream materials and the energy required to manufacture the components were derived from literature data (Ellingsen et al., 2014; Majeau-Bettez et al., 2011; Notter et al., 2010). Table 21 and

Table 22 summarize the information needed for modelling the battery cells, which represent approximately 64% of the total weight (Figure 52).

Figure 51: Battery pack components as clustered for the LCA modelling

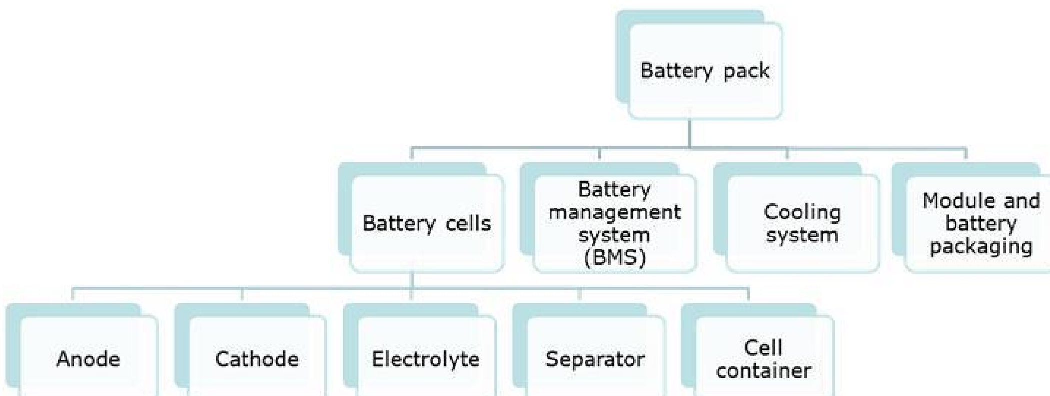



Table 21: Material breakdown of a fresh LMO–NMC/graphite cell as determined by dismantling and further analysis

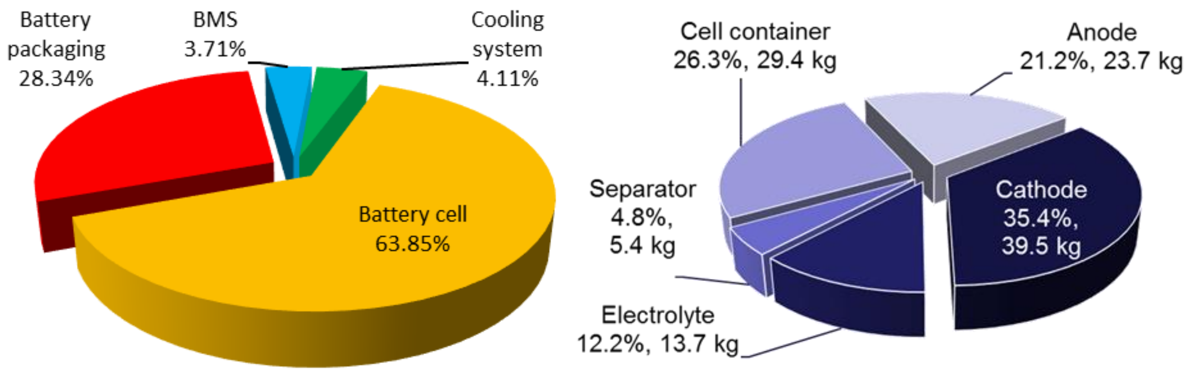
| LMO-NMC cell (total weight before opening: 1396.2 g) | % in weight (%) | Fraction/ g | Accuracy (g) |
|---|---|---|---|
| Steel: external case, connectors | 21.47 | 299.8 | +/−2 |
| Al: current collectors, electrode foils | 3.74 | 52.2 | +/−2 |
| Cu: current collectors, electrode foils | 10.03 | 140.0 | +/−6 |
| Polymer: wrapping, tapes, separator | 5.99 | 83.6 | +/−2 |
| Anode active material: graphite | 10.17 | 142.0 | +/−12 |
| Binder | 2.68 | 37.4 | +/−6 |
| Cathode active material: LMO-NMC | 27.47 | 383.5 | +/−20 |
| Carbon black in the cathode | 3.38 | 47.2 | +/−32 |
| Electrolyte | 13.75 | 192.0 | +/−20 |
| Uncounted materials lost in cutting/drilling/handling (steel, polymer, Cu, Al, active materials) | 1.32 | 18.5 | +/−5 |

Table 22: Bill of Materials of the LMO–NMC cell and main assumptions for cell modelling

| | Composition | Mass (g) |
|---|---|---|
| Anode | | 282.94*** (P) |
| The specific composition of the negative active material and of the binder was unknown, so they were taken from a study (Ellingsen et al., 2014). The required amounts were determined during battery cell dismantling. In anode manufacturing, a solvent was used to give the mixture a slurry texture. After the negative paste was applied to the current collector, the solvent evaporated. The information about solvent is not available, so its composition was modelled in accordance with studies (Ellingsen et al., 2014; Gaines and Cuenca, 2000; Majeau-Bettez et al., 2011). The required amount was taken from Ellingsen et al. (Ellingsen et al., 2014). | Negative current collector: copper (P*) | 113.48 (P) |
| | Negative active material: synthetic graphite (L**) (Ellingsen et al., 2014) | 162.24 (P) |
| | Binder: 0.5 polyacrylic acid (PAA) + 0.5 carboxymethyl cellulose (CMC) (L) | 7.22 (P) |
| | Solvent: N-methyl-2-pyrrolidone (NMP) (L) | 159.8 (L) |
| Cathode | | 502.82*** (P) |
| The specific composition of the positive active material was provided by the battery manufacturer. The active cathode material composition for the analysed battery was modelled as 52% of $LiMn_2O_4$ (LMO) and 48% of $Li(Ni_{0.4}Co_{0.2}Mn_{0.4})O_2$ (NMC). The LMO inventory was taken from the Ecoinvent database, while the NMC inventory was from Majeau-Bettez et al. (2011) and Ellingsen et al. (2014). Based on Ellingsen et al. (2014), the binder was assumed to be PVDF, with the required amounts determined during battery cell dismantling. Similarly to the negative electrode paste, in the positive electrode paste manufacturing NMP was considered to be the solvent and the required amount was taken from Ellingsen et al. (2014). | Positive current collector: aluminium (P) | 40.36 (P) |
| | Positive active material: LMO (P/L) | 217.45 (P) |
| | Positive active material: NMC (P/L) | 200.73 (P) |
| | Binder: polyvinylidene fluoride (PVDF) (L) | 19.68 (P) |
| | Carbon (P) | 24.6 (P) |
| | Solvent: NMP (L) | 189.6 (L) |
| Electrolyte | | 170.58 (P) |
| The specific composition of the electrolyte was not detected during cell dismantling. Therefore, it was modelled in accordance with the literature (Ellingsen et al., 2014; Gaines and Cuenca, 2000; Kim et al., 2016; Notter et al., 2010). The amount of electrolyte per battery cell was determined in the laboratory. | Lithium hexafluorophosphate ($LiPF_6$) (L) | 150.11 (L) |
| | Ethylene carbonate ($C_3H_4O_3$) (L) | 20.47 (L) |
| Separator | | 67.4 (P) |
| The specific material composition of the separator was not determined, so it was modelled in accordance with Nelson et al. (2011). The weight was determined in the laboratory. | Polypropylene, granulate (PP) (L) | 53.92 (L) |
| | Polyethylene, granulate (PE) (L) | 13.48 (L) |
| Cell case | | 372.47 (P) |
| The cell case was made of steel. It contained the anode and cathode soaked with electrolyte and folded together with the separator in two jelly rolls that were properly connected to the two external negative and positive tabs. The composition of the case was obtained by combining the data determined in the laboratory with the LCI by Ellingsen et al. (2014). | Aluminium (P/L) | 11.77 (P) |
| | Copper (P/L) | 26.38 (P) |
| | Packaging film (P/L) | 7.23 (P) |
| | Polyethylene terephthalate, granulate (P/L) | 5.36 (P) |
| | Polypropylene, granulate (PP) (L) | 22 (P) |
| | Steel (P/L) | 299.72 (P) |
| Total | | 1396.20*** |

*Primary data, **Literature data, ***The amounts of NMP used in cathode and anode manufacturing are not included in the total

Figure 52: Mass composition of the battery pack and of the cells by components




Battery components not included in the cell (BMS and the cooling system) are modelled based on literature data. Transport and infrastructure required for the battery components are based on (Ellingsen et al., 2014).

Due to significant discrepancies concerning the electricity mix adopting for the LCA modelling and the amount of energy used for manufacturing the battery, an average value is adopted for the assessment; moreover, a sensitivity analysis is performed of such parameter (Peters and Weil, 2018). For instance, Chinese electricity (employed by Notter et al. (2010)) is significantly less clean than Japanese supply mix (employed by Bauer (2010)). According to the available information, it is assumed that the assembly of the battery occurs in Japan, and thus the Japanese electricity mix, at medium voltage is used. For the amount of energy to manufacture the cells, Ellingsen et al. (2014) presented three possible values for electricity consumption for cell manufacturing: 586 MJ/kWh, 960 MJ/kWh and 2318 MJ/kWh. In this LCA, the average value (960 MJ/kWh) was used.

**Repurposing**

According to (Richa, 2016) and based on analyses of real practices, the repurposing includes the disassembly of the main components of the battery pack (e.g. casing, BMS) down to module level in order to test the state of health (SoH) of the battery (Ahmadi et al., 2014b; Cready et al., 2003). An average transportation of 100 km for the EV battery collection, a disassembly of the battery pack down to module level and one charge/discharge cycle for their testing are considered. According to (Ellingsen et al., 2014) the battery modules are kept together by a battery tray through straps, restraints and foam. For the LCA modelling, it is assumed that a new battery tray should be adopted after the dismantling of the battery pack (Table 23).

Table 23: Data used for the repurposing stage

| Parameter | LMO /NMC Repurposed battery | Source of the information |
|---|---|---|
| Transportation [km] | 100 | Own assumptions |
| Battery tray [kg] | 14.88 | (Ellingsen et al., 2014) |
| Battery retention [kg] | 5.45 | (Ellingsen et al., 2014) |
| Electricity consumption [kWh] | 8.72 | Own assumptions, considering one charge/discharge cycle |

**Use phase**

Consistent with the goal of the study and the method illustrated in section 4.2.1, the use of the battery during its first life (i.e. in PHEV) is not included in the system boundaries of the study. Therefore, the use phase refers to the use of the repurposed battery in the second-use application. To assess the impact of the use of the battery, the flows of energy of the system (application and battery) are considered. Since the energy flows of the system vary according to the considered application, a detailed analysis combining the performance of the battery and the energy requirement of the system are described in separate sections. In particular, the impacts of the adoption of a repurposed LMO/NMC battery to increase the PV self-consumption of a residential house are reported in section 4.3.2, where in section 4.3.3 are reported the modelling of the energy flows of a peak shaving applications and the consequent impacts.

**EoL**

Concerning the EoL, it is assumed that batteries are addressed to recycling, according to the Directive currently in force.

In general, recycling processes can be schematized in mechanical (or physical) processes and metallurgy processes. Disassemble, crushing, screening and separation processes belong to the mechanical processes, while metallurgy processes are chemical processes entailing pyro-, hydro-, bio-metallurgy and combination methods (Yun et al., 2018). Some of the challenges to be faced by the mechanical processes are safety issues (explosion problems, ICBR2019[74]), different design of battery packs and their enclosure in different models of xEVs, and the absence of automatized disassemble processes. Focusing on metallurgy processes, some of the challenges refer to the high energy request (e.g. for pyrometallurgical processes), the potential environmental impacts related to such processes (e.g. impact of used energy or reagents/chemicals), the small flows of available batteries to be treated that influence the economy of scale and the efficiency of recycling processes (Yun et al., 2018).

[74] ICBR Battery Recycling Congress 2018, Berlin, 2018

Therefore, before recycling, the batteries of the battery pack are assumed to be properly collected and sorted (Figure 53, BOX 8). The main components as BMS, cooling system and battery packaging are separated from the cell and treated separately.

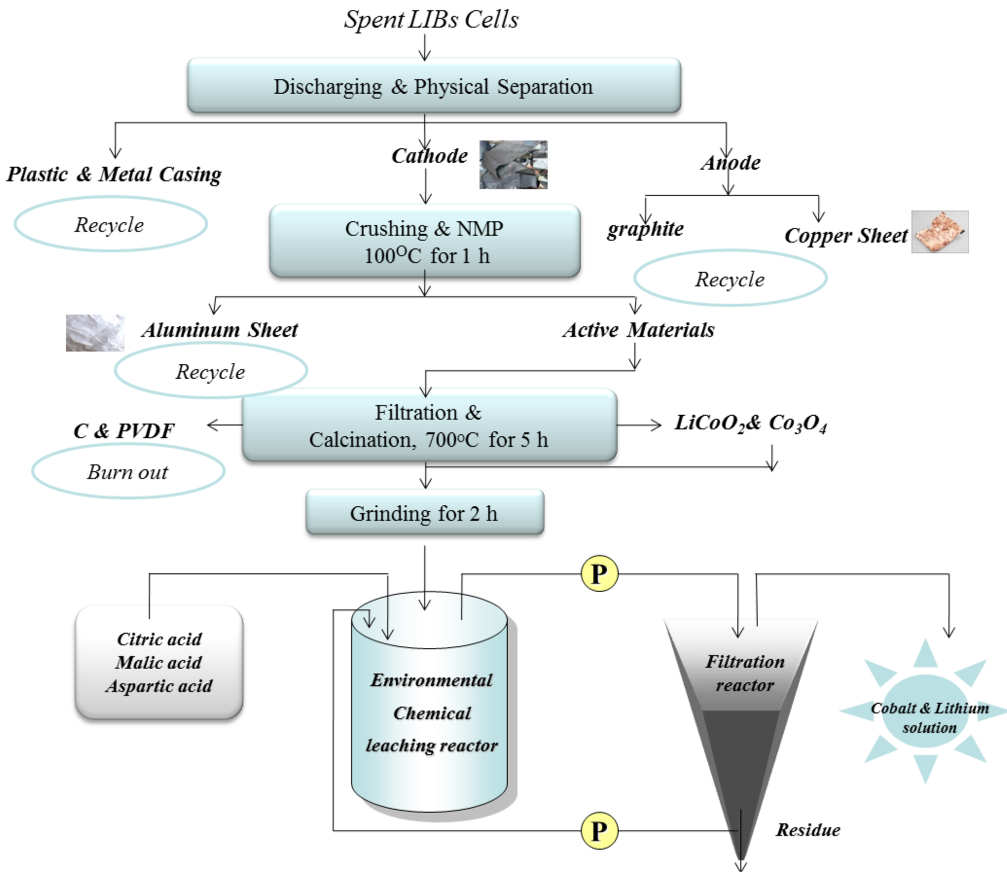
Figure 53: Sorting of batteries at the Van Peperzeel plant (The Netherlands)

According to the literature and the interviewed stakeholders during the research, the most common recycling treatment in Europe is a pyro-metallurgical process, which allow recovering mainly nickel, cobalt, copper and steel[75]. However, hydrometallurgical or combined processes are arising (Friedrich and Peters, 2017; Mathieux et al., 2017) in order to recover much more materials, e.g. metal sulphate, which can be used again to manufacture batteries' active materials (Recharge Association, 2018). The hydrometallurgical process requires the additions of chemicals and additives to recover materials (Joulié et al., 2014; Ordoñez et al., 2016; Vieceli et al., 2018)(Friedrich and Peters, 2017; Ordoñez et al., 2016) besides the higher costs, impacts related to the adoption of such chemicals/additives contribute to the life-cycle impact of the vehicle (Dunn et al., 2012).

In the study, the approach adopted by Richa et al. (2017) is assumed: 50% of pyrometallurgical process and hydrometallurgical process for the reaming 50%. Note that the Ecoinvent dataset was adapted to the specific case study in order to make matching the output and input flows; the amount of material recoverable from the cells are calculated considering the recycling rate reported in (Chancerel et al., 2016).

[75] Valuable materials for which there is already a market, while recovering aluminium, lithium and manganese is not economic or energy efficient to recover (Dunn et al., 2012)

Figure 54: Life-cycle impacts of BEV and ICEV (Dunn et al., 2012)

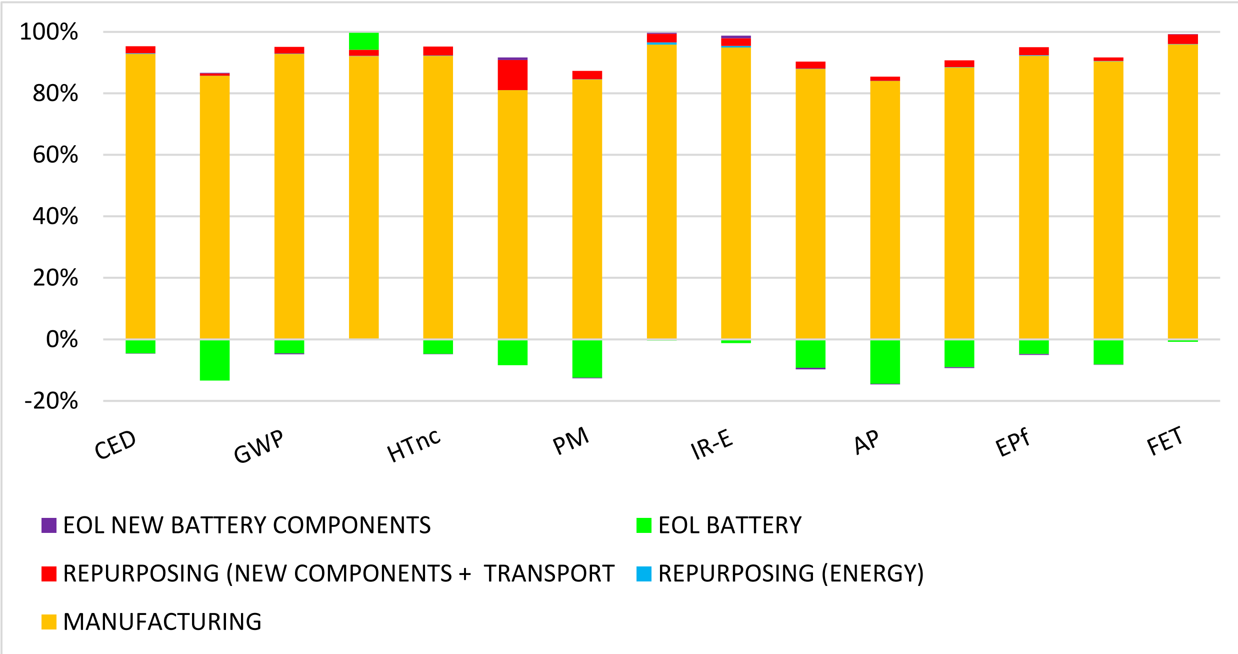



### 4.3.1.3 Life Cycle Impact Assessment (LCIA)

According to the main goal of the work, the LCIA is the background of the environmental assessment of extending the lifetime of traction batteries through their adoption in second-use applications. Therefore, results of the LCIA are hereinafter reported for each life-cycle phase, consistent with the LCI. The impacts are referred to the functional unit of the study, i.e. a LMO/NMC Lithium-ion battery pack for PHEVs as described in section 4.3.1.2.

Figure 55 shows the impact of the manufacturing, the repurposing and the EoL phases. Manufacturing has the major contribution to the impact for all the assessed impact categories and note the EoL.

Figure 55: Battery pack manufacturing (175 kg)




### 4.3.1.4 Life Cycle Interpretation

A more in depth contribution analysis was performed to better understand the contribution of specific components and/or life-cycle phases to the overall impact. Moreover, due to the uncertainty of some input data, a sensitivity analysis of the most important value was performed and published in Cusenza et al. (2019).

**Manufacturing**

From the contribution analysis of the manufacturing phase, it emerged that the energy for manufacturing the battery cells have the highest contribution for almost all the assessed impact categories (Figure 55). Also, the BMS and the packaging impacts are not negligible Focusing on the battery cells, it is noticed that the anode, the cathode and the energy needed for the cells production contribute for more than 70% for all the assessed categories (Figure 57).

Figure 56: Battery pack manufacturing (175 kg) - contribution analysis

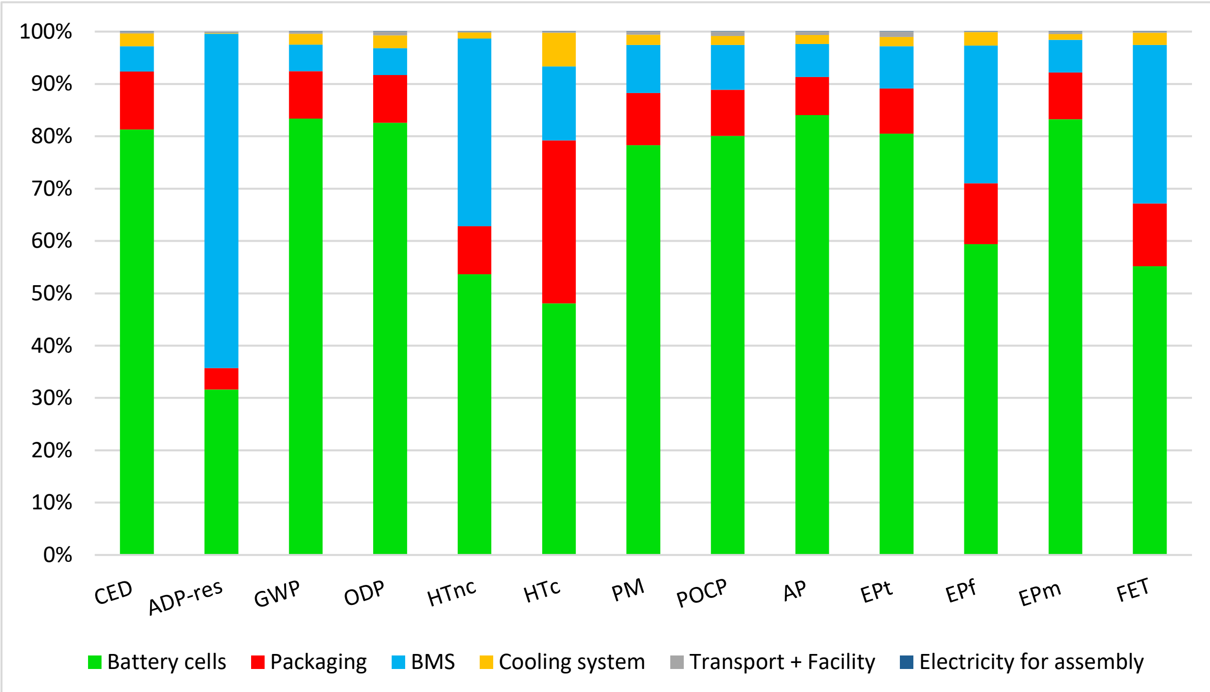
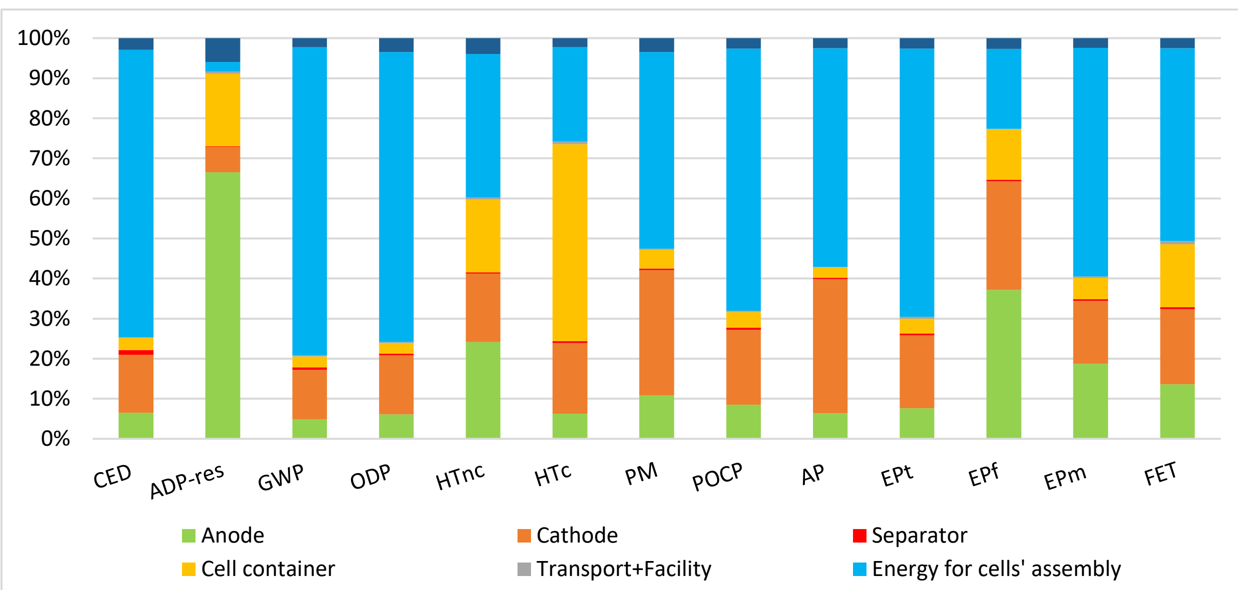



Figure 57: Battery cells manufacturing (111.73 kg) - contribution analysis



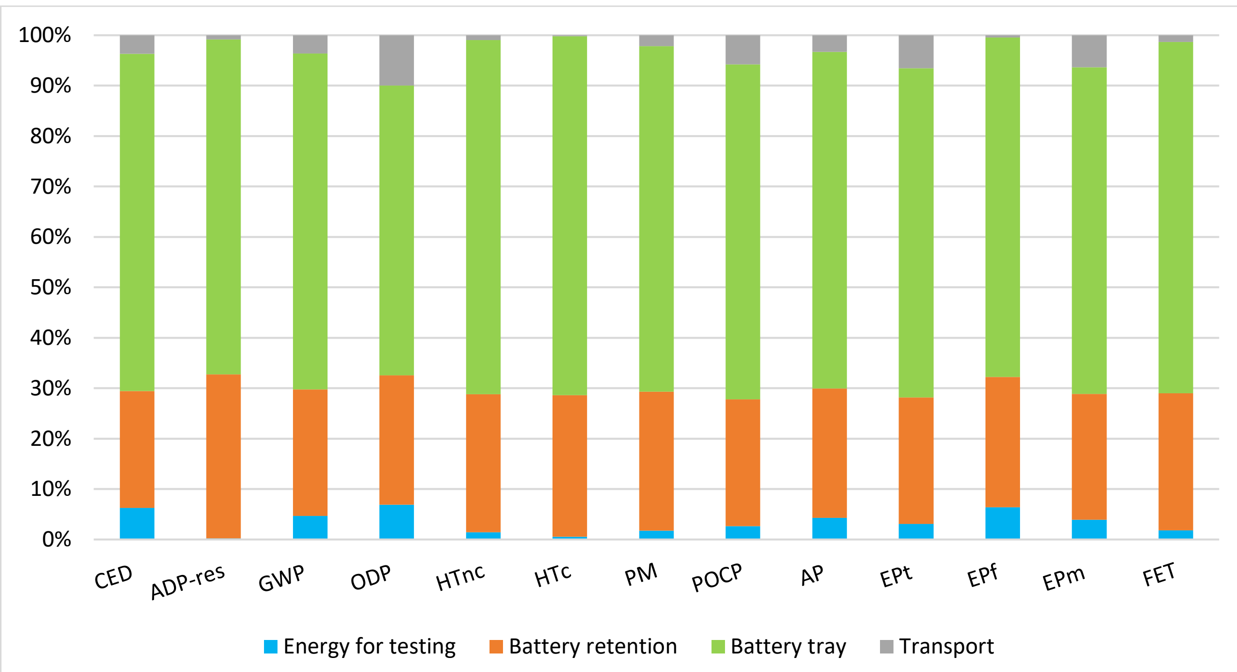
## Repurposing

Results reported in Figure 58 depict the relevance of manufacturing of the battery new components: battery tray and battery retention. Their contribution ranges between 76.60% and 99.25% for respectively the IR and the HT-C impact categories.

The electricity consumed for testing the battery pack is always lower than 21% for all the other assessed impact categories and the contribution of transports could be considered as negligible (maximum contribution for the ODP category, about 10%).

Note that, according to (Cready et al., 2003), 4 charge/discharge cycles are needed for testing the SoH of battery packs whereas only 1 is considered in the LCI (Section 4.3.1.2). However, the performed sensitivity analysis shows that the contribution on the tests will not heavily affect the overall environmental impact. The impact categories mainly affected by this change is the AP (+5.77% of the overall impact), whereas for all the other assessed impact categories the variation can be considered as negligible (it never exceeds 0.70%).

Figure 58: Repurposing stage of the LMO/NMC battery - contribution analysis




## EoL

In Table 24 are reported the overall impacts and the percentage contribution of EoL of the battery pack. The recycling and then the avoided primary production of copper, aluminium and steel determine an avoided impact (i.e. <0) in almost all the impact categories. The only exceptions (positive values, i.e. environmental impacts) are represented by the ODP and FET impact categories (grey cells in Table 24) due to the "sodium hydroxide" used for the pyrometallurgical process and the aluminium in the treatment of the casing.

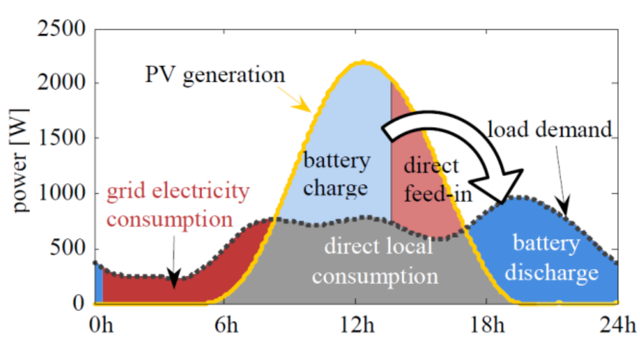
Table 24: Environmental impact assessment of the EoL of one battery pack

| Impact categories | Unit of measure | Treatment of the BMS | Treatment of packaging | Treatment of cooling system | Treatment of battery cells | Total |
|---|---|---|---|---|---|---|
| CED | MJ | -7.91E+01 | -2.94E+03 | -1.02E+03 | 1.34E+03 | -2.69E+03 |
| ADP-res | kg Sb $_{eq}$ | -7.73E-04 | -1.53E-03 | -5.44E-05 | -9.41E-03 | -1.18E-02 |
| GWP | kg CO2 $_{eq}$ | -3.99E+00 | -1.87E+02 | -6.25E+01 | 7.86E+01 | -1.75E+02 |
| ODP | kg CFC-11 $_{eq}$ | -3.63E-07 | -1.11E-05 | -3.73E-06 | 3.13E-05 | 1.61E-05 |
| HTnc | CTUh | -2.10E-06 | -5.48E-05 | -1.37E-05 | -4.36E-05 | -1.14E-04 |
| HTc | CTUh | -1.82E-06 | -2.47E-05 | -5.59E-06 | -7.45E-06 | -3.96E-05 |
| PM | kg PM2.5 $_{eq}$ | -8.18E-03 | -1.68E-01 | -5.10E-02 | -1.40E-01 | -3.67E-01 |
| POCP | kg NMVOC $_{eq}$ | -2.83E-02 | -6.07E-01 | -1.87E-01 | -3.66E-01 | -1.19E+00 |
| AP | molc H+ $_{eq}$ | -5.86E-02 | -1.64E+00 | -5.69E-01 | -3.06E+00 | -5.33E+00 |
| $EP_t$ | molc N $_{eq}$ | -8.99E-02 | -2.00E+00 | -6.45E-01 | -9.89E-01 | -3.73E+00 |
| $EP_f$ | kg P $_{eq}$ | -2.94E-03 | -1.87E-02 | -3.60E-03 | -6.99E-02 | -9.51E-02 |
| $EP_m$ | kg N $_{eq}$ | -2.63E-02 | -2.03E-01 | -5.75E-02 | -2.79E-01 | -5.66E-01 |
| FET | CTUe | 6.76E+01 | -3.16E+02 | -7.58E+01 | -1.50E+02 | -4.74E+02 |

## 4.3.2 Increase of PV self-consumption

For several renewables systems, e.g. photovoltaic systems, the utility consumer does not directly consume a significant amount of the produced energy. Consequently, this energy enters in the grid network or it is lost. One principal approach to balancing an electric power system with a high penetration of time varying renewable resources is energy storage. Through the adoption of a storage battery the surplus of PV energy (i.e. energy not directly consumed by the system) can be stored and used where the PV system could not produce energy (i.e. night) or it could not satisfy the energy demand of the system (Eyer and Corey, 2010). At the European level, PV installations are expected to grow further in the next decade and to play a key role in increasing the local share of renewable energy sources at local level (Taylor et al., 2015). Meanwhile the cost of energy storage is expected to decrease (Bermudez, 2017) and the renewable integration is expected to be one of the most relevant applications of storage batteries (EUROBAT, 2016; Kempener and Borden, 2015).

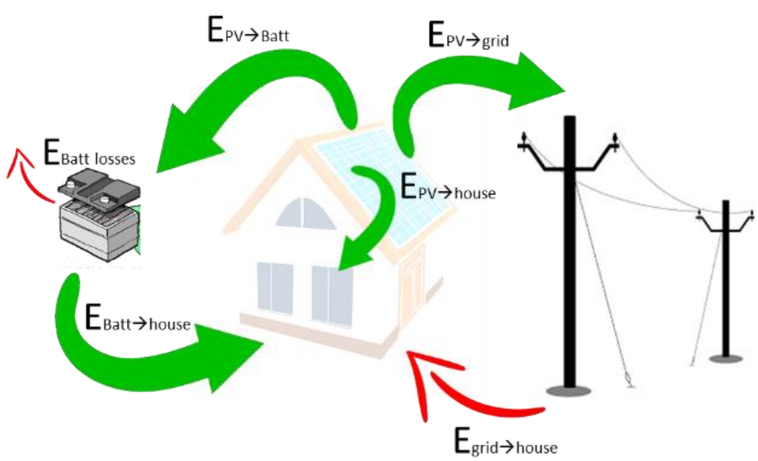
Figure 59: Capacity (Braun and Magnor, 2009)



In Europe, the increase of the PV system in residential and commercial buildings entailed also challenges in predicting power and voltage fluctuations, that can disturb the low voltage grid (e.g. ramps and peaks of injected PV power and hence power reversal, reactive power control) (Aziz and Ketjoy, 2017; Weniger et al., 2014a). Therefore, new policies for managing the PV self-consumption arose in several European Countries and depending on several factors (geographical area and weather conditions, PV penetration level, network characteristics, etc.). An example is the feed-in curtailments, which means the limitation of the feed-in power to a specific value, e.g. 70% (0.7 kW/kWp) in Germany for PV systems below 30 kWp (Aziz and Ketjoy, 2017; Weniger et al., 2014a). However, the Renewable Energy (RES) Directive 2009/28/EC requests the minimization of the use of curtailment, this means the increase of the share of consumer load covered by RES and the decrease of fuel use and generation related emissions of the conventional power plants (Winkler and Regawitz, 2016).

The configuration considered for the environmental assessment of second-use batteries to increase the PV self-consumption in a house is schematized in Figure 60.

Figure 60: Schematic representation of the energy flows of the system



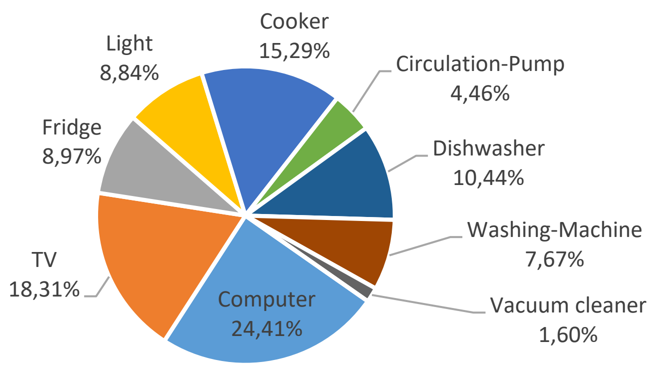

$E_{PV \rightarrow Batt}$ = energy provided to the battery from the PV installation
$E_{PV \rightarrow grid}$ = energy provided to the grid from the PV installation
$E_{Batt\ losses}$ = energy lost due to the battery efficiency
$E_{PV \rightarrow house}$ = energy provided to the house from the PV installation
$E_{Batt \rightarrow house}$ = energy provided to the house from the battery
$E_{grid \rightarrow house}$ = energy required by the house from the grid

$$E_{in} = E_{PV \rightarrow grid} = E_{requirement} - E_{PV \rightarrow house} - E_{PV \rightarrow Batt}$$
$$E_{out} = E_{grid \rightarrow house} = E_{requirement} - E_{PV \rightarrow house} - E_{Batt \rightarrow house}$$

**Input data**

The household load profile is provided by the ResLoadSIM software[76] (time resolution of 1 minute). The system configuration refers to a residential building located in Amsterdam, with 4 residents and a yearly consumption of 5.15E+03 kWh.

Figure 61: Yearly energy consumption of household appliances used for the modelling (Total consumption of the household appliances = 2,140.23 kWh/y) (own elaboration based on ResLoadSIM simulation[77])




Available primary data (15 minutes resolution) for the PV production refer to a real PV installation in a JRC site in The Netherlands[78]. Based on a real case, for the analysis the energy provided by 21 PV panels is considered[79]. Based on (Ciocia, 2017) and on the battery characteristics, the energy flows of the system (schematized in Figure 60) were assessed for one year, every 15 minutes (Figure 62 provides an overview of the size of calculations). Further information on how the capacity model is used to calculate relevant parameters can be found in (Bobba et al., 2018b).

Concluding, it is assumed that a repurposed battery *could potentially replaces a fresh battery storing energy in a house with a PV installation*. The developed method (section 4.2.1) is then applied to this case study. To enlarge the analysis, different configurations are considered (Table 25).

---

[76] https://ses.jrc.ec.europa.eu/our-models-portfolio

[77] Note that these data refer to a typical household in The Netherlands. Among other considerations, it is assumed that all the people in the house have a personal computer and that the vacuum cleaner is used everyday, which is higher than the average considered in section 2.

[78] The system is characterized 2 PV converters connected to 96 modules of 250 W, totalling 24 kWp. The orientation of all the modules is SSE with a slope of 10° (Vandenbergh, 2014).

[79] This evaluation is based on a real case-study for which primary data are being collected.

Figure 62: Overview of the excel calculations of the energy load, the PV production and the data elaboration for estimating the energy flows of the system

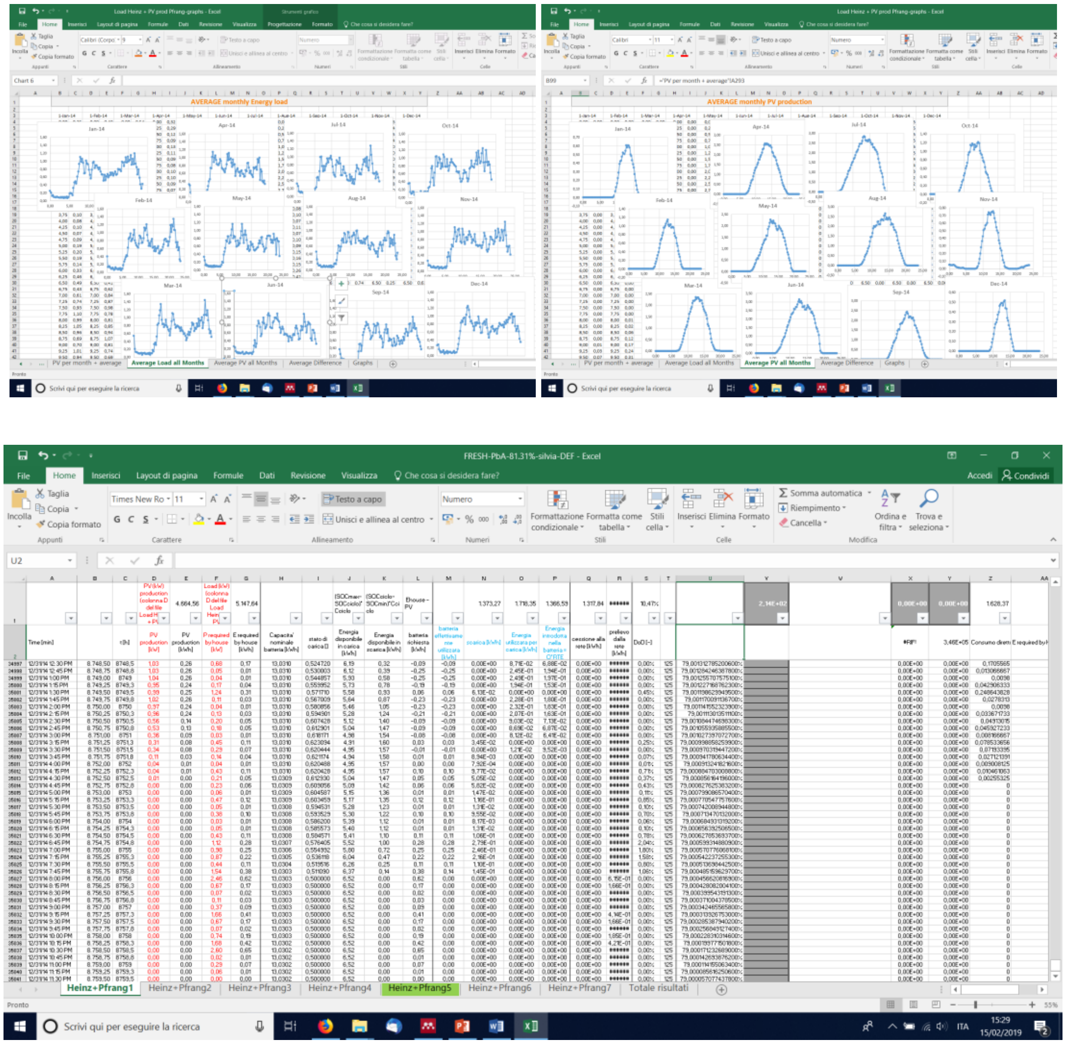



Table 25: Main characteristics of the examined scenarios

| | Reference Scenario | Repurposed Scenario |
|---|---|---|
| Configuration A | Grid connected house<br>PV installation<br>Fresh Li-ion battery storing PV energy | Grid connected house with<br>PV installation<br>Repurposed Li-ion battery storing PV energy |
| Configuration B | Grid connected house<br>PV installation<br>no battery storage system | Grid connected house<br>PV installation<br>Repurposed Li-ion battery storing PV energy |
| Configuration C | Stand - alone house<br>PV installation<br>Diesel-electric generator used to satisfy the energy requirement not satisfied by the PV installation | Stand - alone house<br>PV installation<br>Repurposed Li-ion battery storing PV energy |

In case of no batteries use, according to the literature different feed-in curtailments are considered:

i. no feed-in curtailments are considered;
ii. feed-in curtailments of 70 % kW/kWp are considered;
iii. feed-in curtailments of 50% kW/kWp are considered.

The above mentioned model was applied and the lifetime of both fresh and repurposed batteries was calculated. Results are reported in Table 26.

After about 4 years, 1 repurposed battery is no longer able to satisfy the energy requirement of the house, due to a capacity lower than 60% of the nominal capacity. If a fresh battery is used in the same system, its nominal capacity decreases until 60% of the nominal capacity of the battery after 7 years.

Table 26: Energy flows for the different configurations and the corresponding battery lifetimes

| **Parameter** | **Repurposed Scenario** | **Reference scenario - A** | **Reference scenario - B.i** | **Reference scenario - B.ii** | **Reference scenario - B.iii** | **Reference scenario - C** |
|---|---|---|---|---|---|---|
| Lifetime [ year] | 3.6 | 7.4 | 1 | 1 | 1 | 1 |
| Electricity required by house [kWh] | 1.85E+04 | 3.81E+04 | 5.15E+03 | 5.15E+03 | 5.15E+03 | 5.15E+03 |
| Direct electricity consumption from PV [kWh] - $E_{PV \rightarrow house}$ | 6.02E+03 | 1.24E+04 | 1.68E+03 | 1.68E+03 | 1.68E+03 | 1.68E+03 |
| Electricity provided by batteries [kWh] - $E_{Batt \rightarrow house}$ | 5.14E+03 | 1.11E+04 | 0.00E+00 | 0.00E+00 | 0.00E+00 | 0.00E+00 |
| Electricity needed for charging batteries [kWh] - $E_{PV \rightarrow Batt}$ | 5.51E+03 | 1.17E+04 | 0.00E+00 | 0.00E+00 | 0.00E+00 | 0.00E+00 |
| Electricity from the grid [kWh] - $E_{out}$ = $E_{grid \rightarrow house}$ | 7.29E+03 | 1.46E+04 | 3.47E+03 | 3.47E+03 | 3.47E+03 | 3.47E+03 |
| PV production [kWh] | 1.73E+04 | 3.57E+04 | 4.83E+03 | 4.83E+03 | 4.83E+03 | 4.83E+03 |
| Electricity potentially to be fed in the grid [kWh] - $E_{in}$ = $E_{PV \rightarrow grid}$ | 5.78E+03 | 1.16E+04 | 3.15E+03 | 3.15E+03 | 3.15E+03 | 0.00E+00 |
| Energy losses due to fee-in curtailments [kWh] | --- | | 0.00E+00 | 3.24E+00 | 1.66E+02 | --- |

Note that two different allocation factors, $\alpha = \beta = 0$ (case B1) and $\alpha = \beta = 0.25$ (case B2) are considered for the assessment.

Table 27 summarizes the input data used for the environmental assessment of second-use of the LMO/NMC battery to increase the VV self-consumption of a house. Note that α = β = 0, i.e. no impacts of battery's manufacturing and EoL are allocated to the second use application

Table 27: Summary of the data used for the calculation of the difference between different scenario ($\Delta_{reuse}$)

| Parameter | CED | ADP | GWP | HTc |
|---|---|---|---|---|
| $\alpha^{+}$ | 0 | 0 | 0 | 0 |
| $P_{EVB}$ | 5.51E+04 | 7.57E-02 | 3.56E+03 | 3.85E-04 |
| $Rep_{EVB}$ | 1.48E+03 | 6.92E-04 | 8.81E+01 | 4.68E-05 |
| $\beta^{+}$ | 0 | 0 | 0 | 0 |
| $E_{EVB}$ | -2.69E+03 | -1.18E-02 | -1.75E+02 | -3.96E-05 |
| $E_{EVB\ new\ components}$ | -5.40E+01 | 1.58E-04 | -1.14E+01 | 3.64E-06 |
| $T_{EVB-stor}$ | 3.6 | 3.6 | 3.6 | 3.6 |
| $P_{B^{*}-stor}$ | 5.51E+04 | 7.57E-02 | 3.56E+03 | 3.85E-04 |
| $E_{B^{*}-stor}$ | -2.69E+03 | -1.18E-02 | -1.75E+02 | -3.96E-05 |
| $T_{B^{*}-stor}$ | 7.4 | 7.4 | 7.4 | 7.4 |
| | **REPURPOSED SCENARIO*** | | | |
| $U'_{EVB-stor,n}$ (grid-connected house) | 1.34E+04 | 1.02E-03 | 7.70E+02 | 2.67E-05 |
| $U'_{EVB-stor,n}$ (stand-alone house) | 1.04E+05 | 8.56E-03 | 6.88E+03 | 1.20E-04 |
| | **REFERENCE SCENARIO – A* (grid-connected house)** | | | |
| $U_{B^{*}-stor,n}$ | 2.79E+04 | 2.12E-03 | 1.60E+03 | 5.56E-05 |
| | **REFERENCE SCENARIO – B* (grid-connected house)** | | | |
| $U_{B^{*}-stor,n}$ | 2.82E+03 | 2.14E-04 | 1.62E+02 | 5.62E-06 |
| | **REFERENCE SCENARIO – C* (stand-alone house)** | | | |
| $U_{B^{*}-stor,n}$ | 4.96E+04 | 4.07E-03 | 3.27E+03 | 5.71E-05 |

+ according to Section 3.4.1, for this study the environmental impact of EV battery manufacturing and EoL is fully allocated to the first life (i.e. α=β=0). The performed sensitivity analysis considers other values (Section 4.4.1)

*Use stage impacts (U) refer to the difference between the electricity input and the output of the system as stated in Section 4.2.1.3(i.e.$E_{grid/generator \rightarrow house} - E_{PV \rightarrow grid}$).

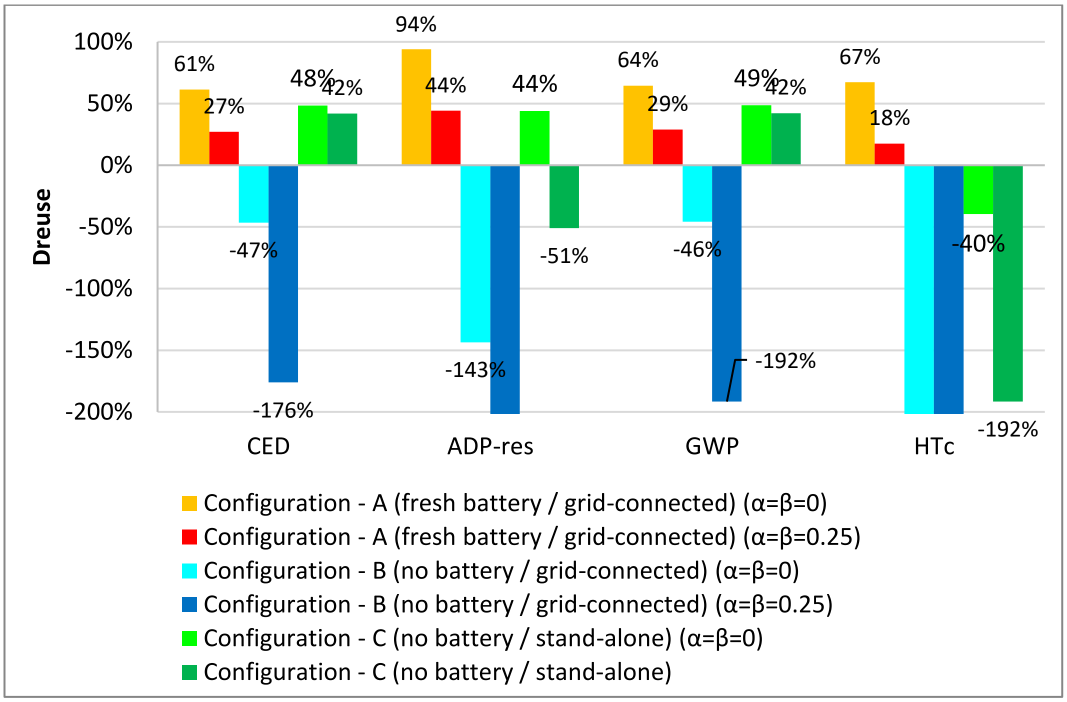
**<u>Analysis results</u>**

The difference between the two scenarios ($\Delta_{reuse}$) was calculated for the all the impact categories presented illustrated in section 4.3.1.1. Focusing on 4 representative impact categories, the resulting indexes for four representative impact categories are showed in Figure 63. It is observed that:

- the replacement of a fresh LMO/NMC battery with a repurposed EV battery having residual capacity of 81.31% after its first life (Configuration– A) is beneficial for all the impact categories, i.e. $D_{reuse} > 0$. Moreover, environmental benefits are higher for the impact categories mainly affected by the manufacturing stage (i.e. ADP-res). In detail, the use of a repurposed battery in a grid-connected house to increase the PV self-consumption allows the reduction of 94% of the life cycle ADP-res compared to a Reference Scenario in which a fresh battery is used in a grid-connected house;
- in case of the adoption of a repurposed battery in a grid-connected house without any batteries to be replaced (Configuration – B) environmental drawbacks are observed for all the assessed impact categories. For instance, even if the adoption of the repurposed battery allows to maximize the consumption of local electricity, the life-cycle GWP is increased of 46% compared to the life-cycle GWP on the Reference Scenario. This is mainly due to the repurposing of the EV battery, the need of new battery components and the energy losses due to the battery efficiency;

- in stand-alone houses with a diesel generator (Configuration – C), the adoption of a repurposed battery shows benefits for all the assessed impact categories even if the repurposed battery does not substitute a fresh battery. In detail, a life cycle GWP reduction (49%) is observed if a repurposed battery is used in a stand-alone house using a generator for its energy requirement.

Figure 63: Index assessing the energy and environmental assessments of adopting a repurposed EV battery to increase a house's PV self-consumption.




Since the impact of repurposing was modelled through secondary data and assumptions (section 4.3.1.2), and there are different actions that could occur in repurposing, a sensitivity analysis was performed considering two different assumptions:

1. the energy need for the repurposing is considered to be 4 times higher the energy considered for the initial LCA modelling, while the impact of the replaced components is increased by 0.5 times;
2. the energy need for the repurposing is considered to be 4 times higher the energy considered for the initial LCA modelling; moreover, it is assumed that 4 cells have been replaced in the battery.

Results are reported in Figure 64 and Figure 65. The durability index decreases compared to the index depicted in Figure 63 even though the difference for configurations A is not significant. For Configuration B (i.e. the battery is adopted without replacing any fresh battery, the increase of the impact of repurposing correspond to very low value of the index. Note that also that also the impact of the EoL of the replaced cell has a not negligible contribution to the environmental impact.

Figure 64: Index assessing the energy and environmental assessments of adopting a repurposed EV battery to increase a house's PV self-consumption. Increased of the energy needed for testing

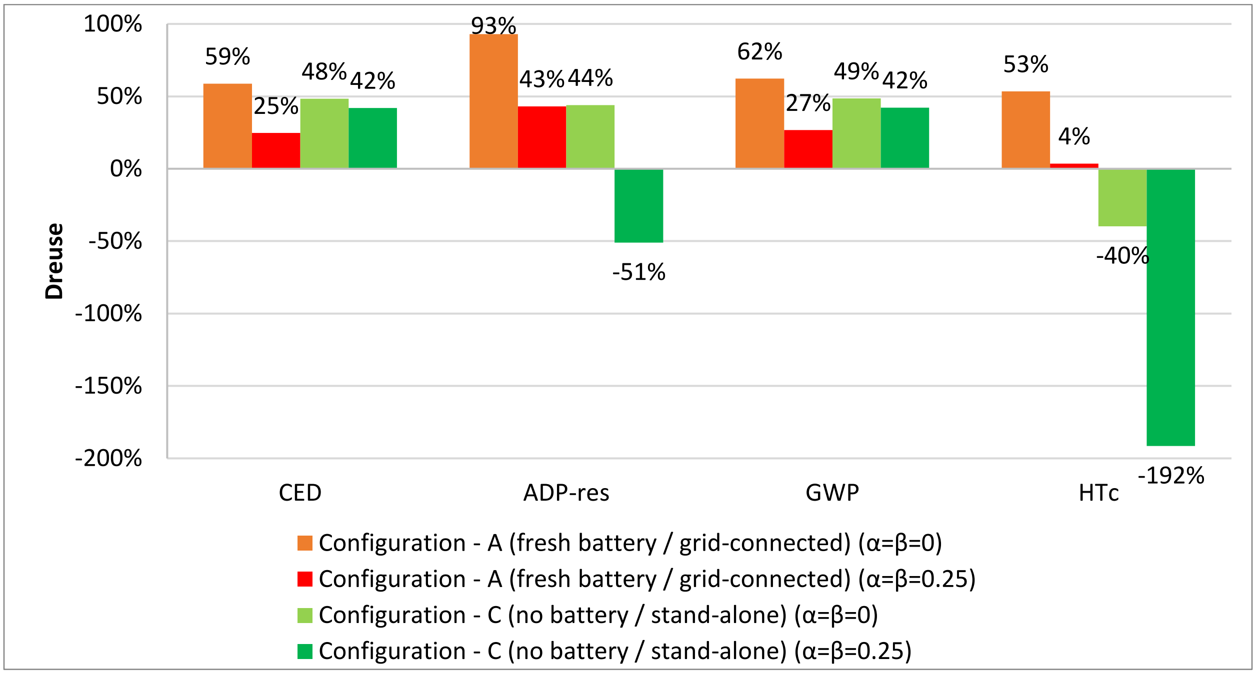
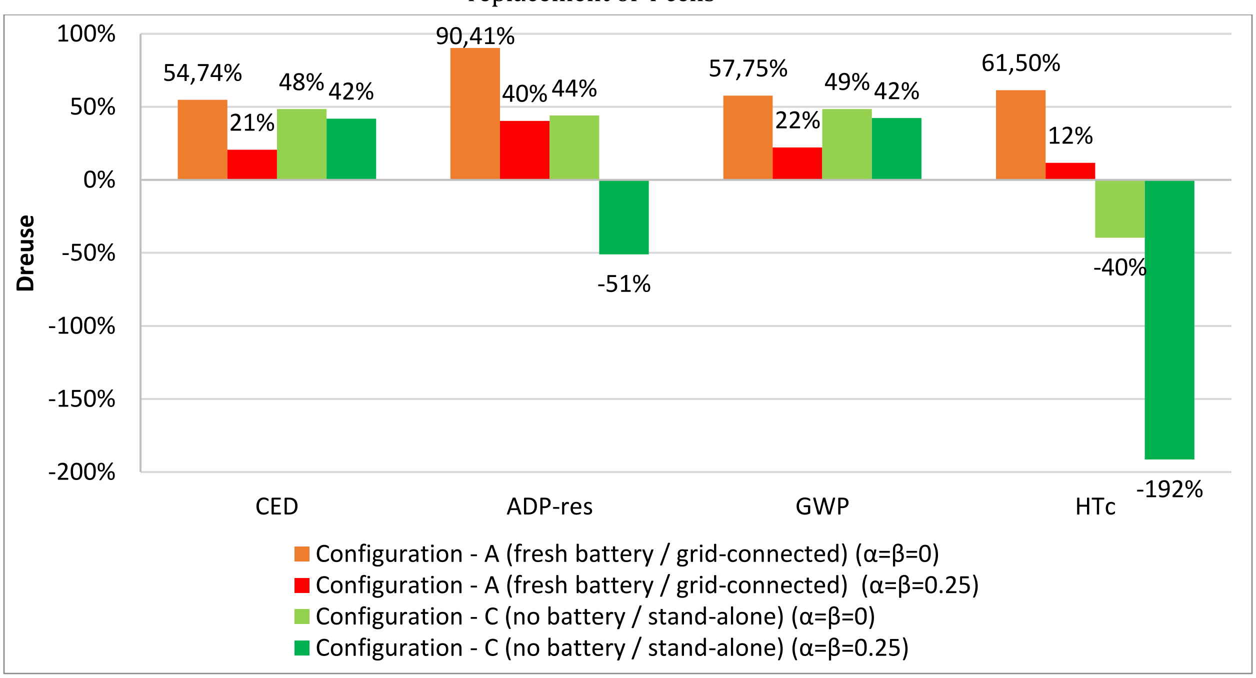



Figure 65: Index assessing the energy and environmental assessments of adopting a repurposed EV battery to increase a house's PV self-consumption. Increased of the energy needed for testing and replacement of 4 cells

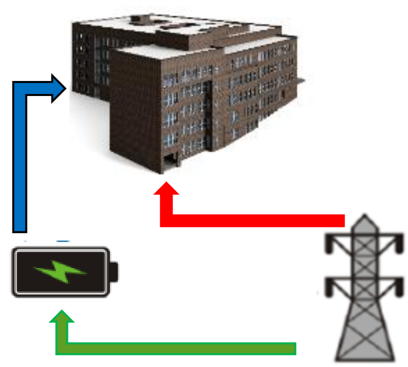


### 4.3.3 Peak shaving

This is an energy storage application that requires discharge of the ESS during the daily on-peak period for electric power (duration on the order of 2 to 12 hours) and is intended to recharge in the daily off-peak period for electric power (Schoenwald and Ellison, 2016). The peak shaving service can be used for shifting electricity demand to relieve peak demand charges, thus ensuring a saving for the customers. Also, the peak shaving service can be used to increase the self-consumption of renewable energy. In this case, the PV energy that is exceeding the permitted feed-in limit is stored in the battery avoiding the loss of such energy (Litjens et al., 2016; Weniger et al., 2014).

This application requires one charge/discharge cycle per day (Bray et al., 2012).

Figure 66: Schematic representation of the peak shaving applications

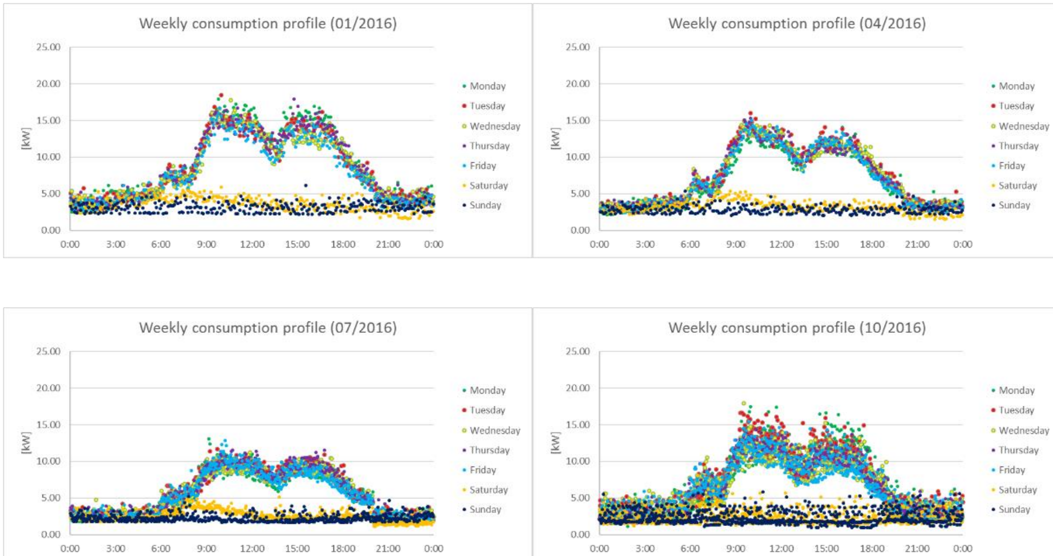


**Input data**

The analysed system is an office building at JRC - Ispra (Building 6) with a total area of 1,444 $m^2$, a volume of 4,706 $m^3$ without any PV system and without any lab area. Therefore, energy consumption is related only to offices. For the environmental assessment, the input/output energy flows are calculated according to the battery's and the system's characteristics (note that the energy delivered by the batteries is covering the peak during the day while batteries are charged during the night).

Data of the daily consumption profile of the building were available on yearly base with 5 minutes resolution. Data of 4 representative months are processed to obtain the average energy requirement for each season (January for winter, April for spring, July for summer and October for autumn). Results of the analysis show that the maximum peak occur during winter (23.16 kW) (Figure 67).

Figure 67: Average daily load profile of the assessed building for four season (own elaboration)




Considering the load profile of the worst day was considered (Wednesdays during winter) and an assumed contracted power of 8 kW, the peak to be shaved is calculated for each representative month. Since data refer to one month per season, the maximum energy requirement is increased of 10% in order to oversize the battery system and be sure to cover all the peaks. According to the characteristics of the repurposed battery (Table 20), the number of batteries needed are calculated considering the following equation[80]:

$$N_b = \frac{\text{Max peak to be shaved in winter season increased of 10\% (worst case) [kWh]}}{\text{Retained capacity after first use [kWh]} \times \text{DoD}}$$

$$= \frac{55.437 \text{ [kWh]}}{9.27 \text{ [kWh]} \times 75\%} = 8$$

The energy required for charging the batteries is calculated using the following equation:

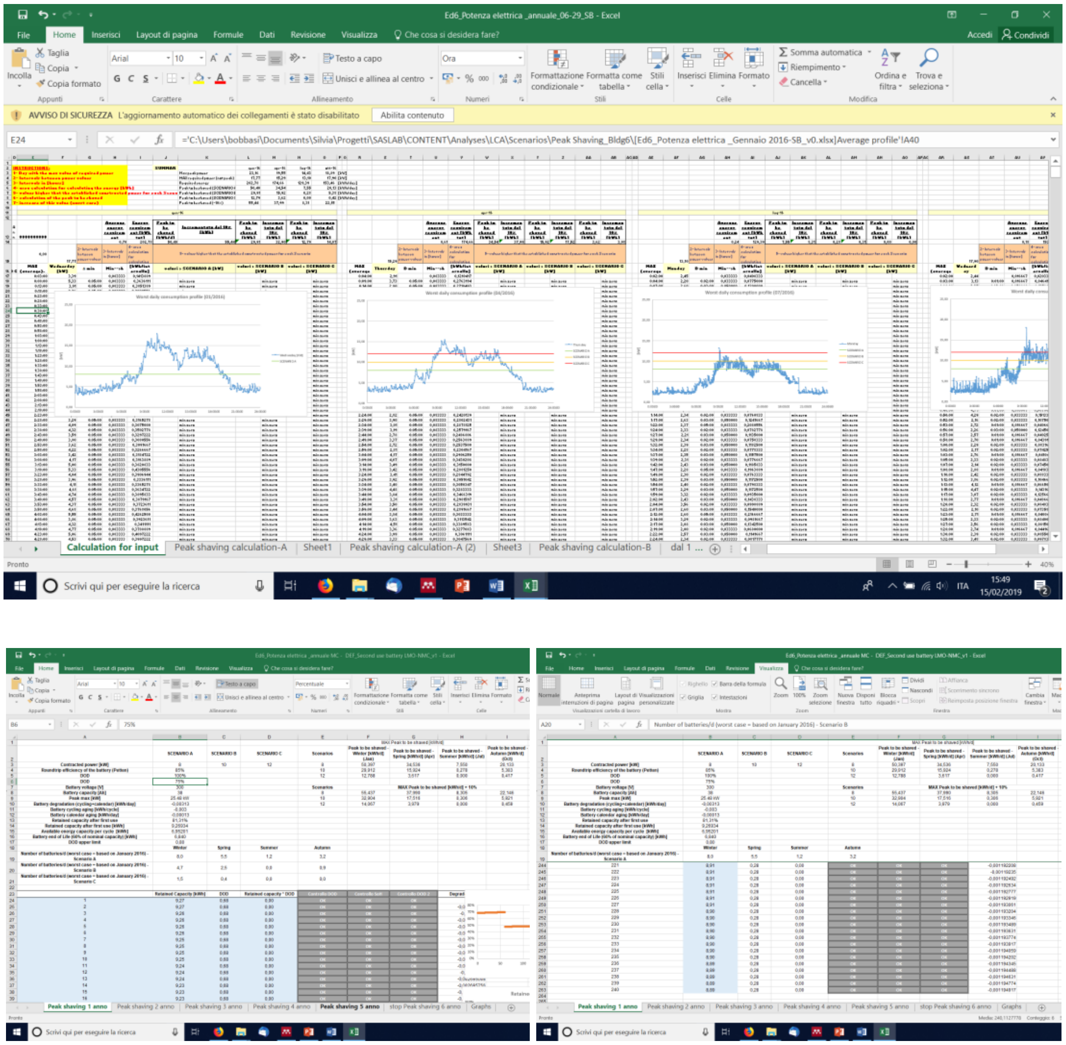
$$\text{Energy required for charging} = \frac{\text{Delivered (Discharge) energy}}{\text{Roundtrip efficiency}} \text{ [kWh]}$$

$$= \frac{\text{Retained capacity } \times \text{DoD}}{\text{Roundtrip efficiency}} \text{ [kWh]}$$

In Figure 68, an overview of the calculation of the energy load and the data elaboration for estimating the energy flows of the system is provided.

[80] For initial calculations, a DoD equal to 75% is used according to consulted experts

Figure 68: Overview of the excel calculations of the energy load and the data elaboration for estimating the energy flows of the system



In addition, in this case, different configurations are considered (Table 29.

Table 28). The above-mentioned model was applied and the lifetime of both fresh and repurposed batteries was calculated. Results are reported in Table 29.

Table 28: Main characteristics of the examined scenarios

| | Reference Scenario | Repurposed Scenario |
|---|---|---|
| Configuration A | Office building located in Ispra (IT)<br>Fresh Li-ion battery storing NG energy | Office building located in Ispra (IT)<br>Repurposed Li-ion battery storing NG energy |
| Configuration B | Office building located in Ispra (IT)<br>no battery storage system | Office building located in Ispra (IT)<br>Repurposed Li-ion battery storing NG energy |

Table 29: System energy requirements

| | January (winter) | April (spring) | July (summer) | October (autumn) |
|---|---|---|---|---|
| Max peak power [kW] | 23.16 | 19.55 | 14.43 | 18.89 |
| Required energy [kWh/day] | 202.78 | 174.66 | 129.39 | 153.46 |
| Peak to be shaved [kWh/day] | 50.40 | 34.54 | 7.55 | 20.13 |
| Peak to be shaved (+10%) [kWh/day] | 55.44 | 37.99 | 8.31 | 22.15 |

Table 30 summarizes the input data used for the environmental assessment of second use of the LMO/NMC battery for the peak shaving in an office building. Note that α = β = 0, i.e. no impacts of battery's manufacturing and EoL are allocated to the second use application.

Table 30: Summary of the data used for the calculation of the difference between different scenario ($\Delta_{reuse}$)

| Parameter | CED | ADP | GWP | HTc |
|---|---|---|---|---|
| $\alpha^{+}$ | 0 | 0 | 0 | 0 |
| $P_{EVB}$ | 5.51E+04 | 7.57E-02 | 3.56E+03 | 3.85E-04 |
| $Rep_{EVB}$ | 1.48E+03 | 6.92E-04 | 8.81E+01 | 4.68E-05 |
| $\beta^{+}$ | 0 | 0 | 0 | 0 |
| $E_{EVB}$ | -2.69E+03 | -1.18E-02 | -1.75E+02 | -3.96E-05 |
| $E_{EVB\ new\ components}$ | -5.40E+01 | 1.58E-04 | -1.14E+01 | 3.64E-06 |
| $T_{EVB-stor}$ | 4 | 4 | 4 | 4 |
| $P_{B^{*}-stor}$ | 5.51E+04 | 7.57E-02 | 3.56E+03 | 3.85E-04 |
| $E_{B^{*}-stor}$ | -2.69E+03 | -1.18E-02 | -1.75E+02 | -3.96E-05 |
| $T_{B^{*}-stor}$ | 6 | 6 | 6 | 6 |
| | **REPURPOSED SCENARIO*** | | | |
| $U'_{EVB-stor,n}$ (grid-connected house) | 1.35E+06 | 1.03E-01 | 7.76E+04 | 6.29E-04 |
| | **REFERENCE SCENARIO – A* (grid-connected house)** | | | |
| $U_{B^{*}-stor,n}$ | 2.17E+06 | 1.78E-01 | 1.43E+05 | 6.02E-04 |
| | **REFERENCE SCENARIO – B* (no battery)** | | | |
| $U_{B^{*}-stor,n}$ | 1.41E+06 | 1.07E-01 | 8.11E+04 | 6.57E-04 |

+ according to Section 3.4.1, for this study the environmental impact of EV battery manufacturing and EoL is fully allocated to the first life (i.e. α=β=0). The performed sensitivity analysis considers other values (Section 4.4.1)

*Use stage impacts (U) refer to the difference between the electricity input and the output of the system (i.e.$E_{grid/generator\rightarrow house} - E_{PV\rightarrow grid}$).

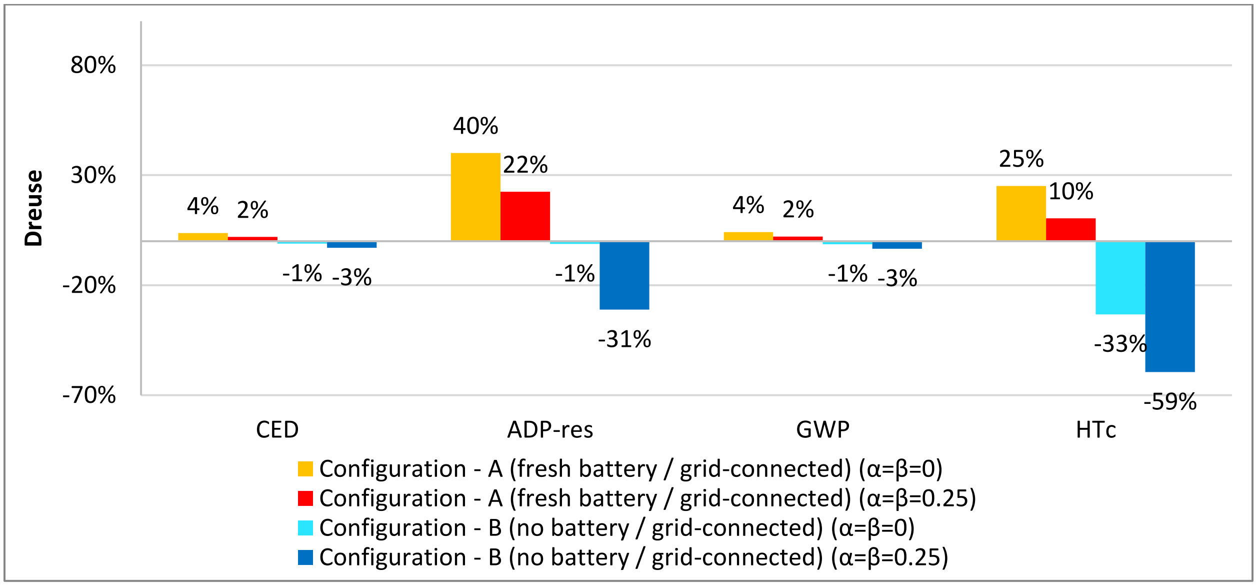
**<u>Analysis results</u>**

A for the PV self-consumption applications, the difference between the two scenarios ($\Delta_{reuse}$) was calculated for the all the impact categories presented illustrated in section 4.3.1.1. Focusing on 4 representative impact categories, the resulting indexes for four representative impact categories are sown in Figure 69. From the results, it emerged that the replacement of a fresh LMO/NMC battery with a repurposed EV battery having residual capacity of 81.31% after its first life (Configuration–A) is slightly beneficial for all the impact categories, i.e. $D_{reuse} > 0$. It is noticed that higher benefits correspond to the ADP-res and the HTc impact categories. Moreover, if the batteries are adopted in a building without replacing any fresh Li-ion battery, the $D_{reuse}$ is negative, that means that there are no environmental benefits in this configuration. In case of CED and GWP impact categories the $D_{reuse}$ is lower than ± 5 % for all the assessed configurations.

Note that the energy mix used in the assessment, in particular to the feedstock providing the energy during the peak hours affects results of the LCA. In specific Countries, where differences in feedstock are relevant, this is a relevant aspect to be assessed in the LCA (Messagie et al., 2014c).

Overall, compared to the $D_{reuse}$ of the application in section 4.3.2, in this case the $D_{reuse}$ is quite lower, therefore the potential environmental benefits are lower than in the increase of pV self-consumption.

Figure 69: Index assessing the energy and environmental assessments of adopting a repurposed EV battery to increase a house's PV self-consumption




Similarly as for the increase of peak shaving, a sensitivity analysis was performed considering two different assumptions:

1. the energy need for the repurposing is considered to be 4 times higher the energy considered for the initial LCA modelling, while the impact of the replaced components is increased by 0.5 times;
2. the energy need for the repurposing is considered to be 4 times higher the energy considered for the initial LCA modelling; moreover, it is assumed that 4 cells have been replaced in the battery.

Results are reported in Figure 70 and Figure 71. The durability index slightly decreases even though differences are not significant on the overall impact. Highest changes are related to the HTc and ADP-res impact categories, which are mainly affected by repurposing.

Figure 70: Index assessing the energy and environmental assessments of adopting a repurposed EV battery to a peak shaving applications. Increased of the energy needed for testing

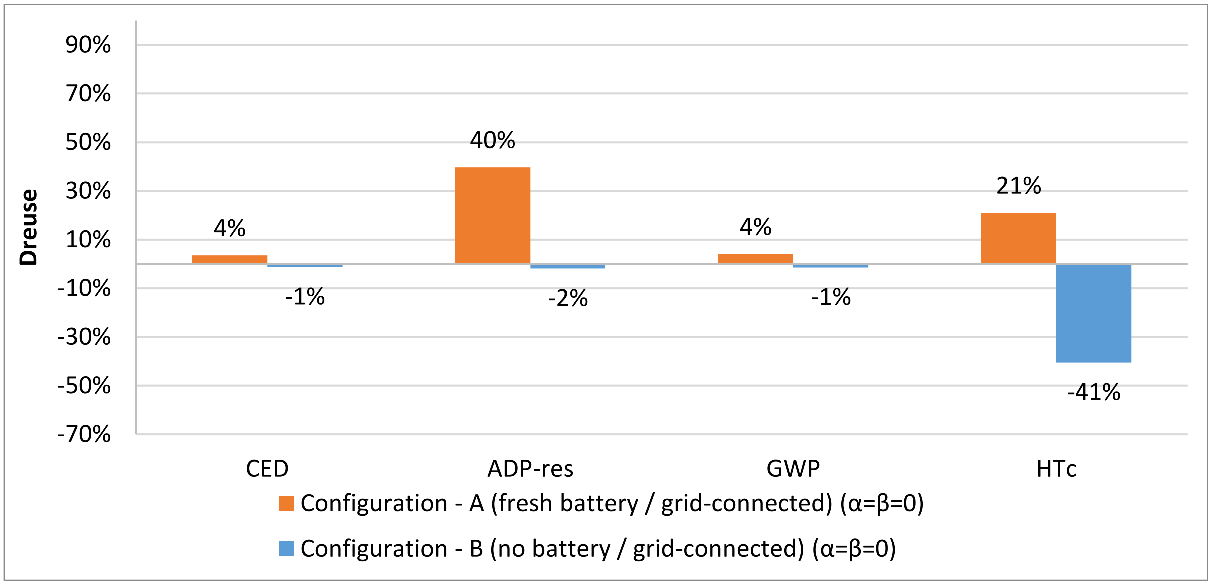
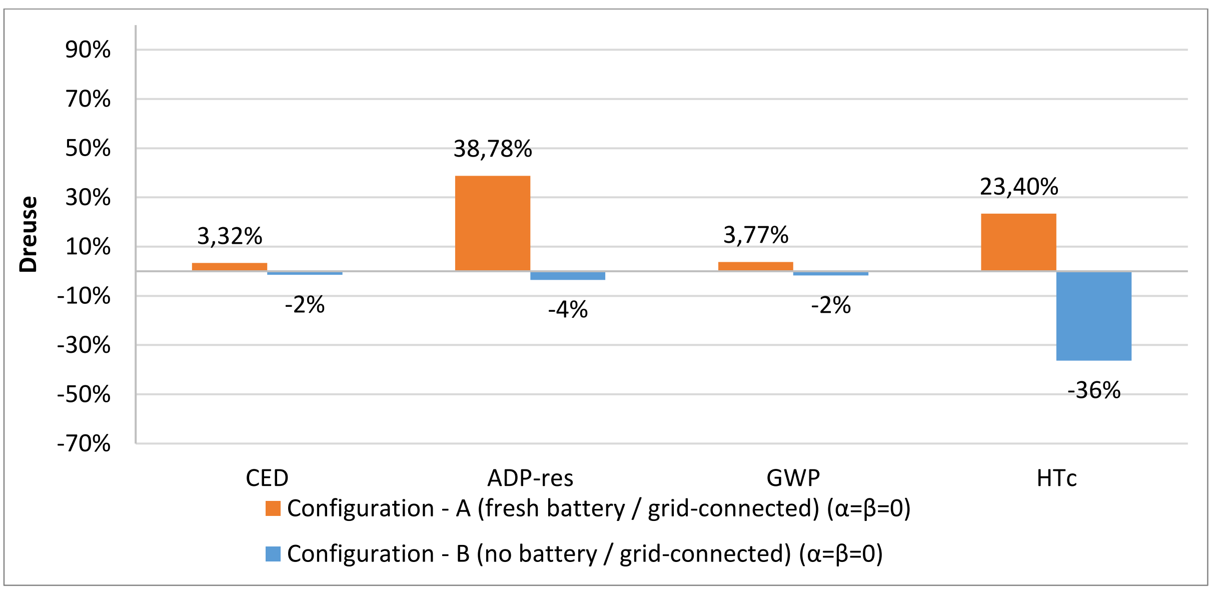



Figure 71: Index assessing the energy and environmental assessments of adopting a repurposed EV battery to a peak shaving applications. Increased of the energy needed for testing and replacement of 4 cells



## 4.4 Conclusions and follow up

The environmental assessment of extending the lifetime of traction batteries was performed through the development of an adapted LCA and then applied to different case-studies.

As first step of the analysis an LCA on an LMO/NMC battery was performed. To model the impacts of the battery manufacturing, both primary[81] and secondary data were used. Secondary data are used to complement the LCA modelling.
This data were used as input for assessing the potential benefits/drawbacks (in a life-cycle perspective) of extending the lifetime of such a battery is two different second-use applications, for which primary data were available: peak shaving and increase of PV self-consumption. Due to the relevance of the use phase in the LCA, the impacts of this phase were modelled separately and through the analysis of the energy flows of the two systems. Energy modelling were developed ad hoc for the specific applications.
The developed model allow comparing different systems and configurations. Therefore, for ach applications different aspects were included in the assessment (e.g. system configuration, battery chemistry, etc.)

Results of the analysis depict that there are environmental benefits is adopting a repurposed LMO/NMC battery in place of a fresh LMO/NMC battery in both applications. Higher benefits are related to the increase the PV self-consumption of a residential dwelling. This is mainly due to the avoided battery manufacturing (in case of fresh LMO/NMC battery). Moreover, the adoption of a repurposed battery in place of a PbA battery also show some benefits due to the higher performance of the Li-ion battery. Note that for the modelling of the PbA batter secondary data were used and a further in-depth analysis is required to confirm results.
In case on peak shaving application in a grid-connected office building in Italy, the adoption of a repurposed battery in a building in which no batteries were previously used does not entail benefits.

Overall, second-use can entail environmental benefits according to the specificities of the assessed system (i.e. battery + application). In performing the assessment, some relevant aspect emerged:

- Relevance of the use phase: detailed model of the use stage are needed to have robust and reliable results of the analysis; the clear understanding of this stage depends on the characteristics of both the system and the battery, and their relation. The modelling of the real energy flow of the system could offer a better understanding of the system and real data could offer a more realistic overview of the real benefits related to the adoption of repurposed batteries
- Differences between batteries performances and their history (i.e. how they are used in their first life) could heavily affect the life-cycle impacts, for instance to battery efficiency, battery (nominal/residual) capacity, etc. Batteries have different performances according to their

[81] The cells dismantling in the JRC Petten laboratories provided the bill of material used to model the environmental impact of the LMO/NMC battery cells

first life, e.g. the energy density of batteries to be used in BEV is higher than the energy density of batteries for PHEV. Their capacity, at the end of the first life is also different and it should be considered when assessing the suitability of such a battery in a specific second-use application.

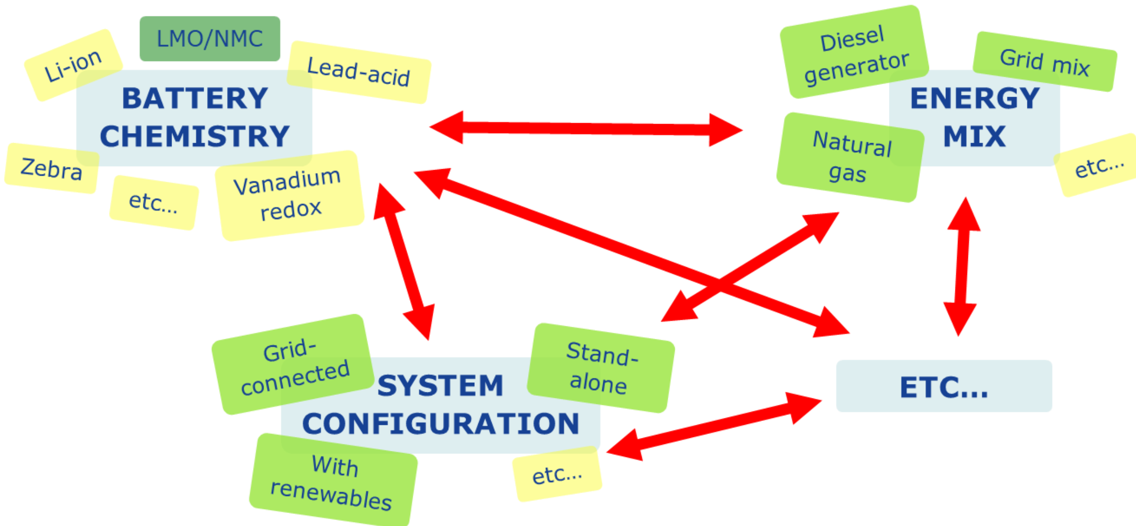
- Battery chemistry should be considered in the assessment as it affects both the manufacturing phase and the energy performances of the battery along its life. Moreover, the fast development of the batteries' technology should be considered when assessing the impacts of batteries second-use [82].
- Lack of data increase the uncertainty of the analysis. Since in both the assessed applications, the impacts of the battery manufacturing, repurposing and EoL are not negligible for all the assessed impact categories, an accurate data collection, possibly in collaboration with stakeholders of the batteries' value chain should be improved. This will allow basing the assessment of robust and reliable data.
- From a methodological perspective, allocation of the impacts of manufacturing and EoL stages along the first and the second life of products is still an open issue and several approaches coexist in the scientific literature. Parameters introduced in the adapted LCA allow assessing the relevance of allocation in the life-cycle impacts. Note that the creation of a market for second-use applications of xEV batteries could affect the choice of the value of these allocation factors and, consequently, the environmental benefits/drawbacks of second-use of xEV batteries

Figure 72 illustrates how the model could be used to enlarge the analyses of second-using traction batteries. The specificity of the assessment is a relevant aspect and the performance of analyses for different types of chemistries used in various applications[83] and in different places can provide a better overview of the potential benefits that second-use of traction batteries can provide in Europe.

[82] New technologies are expected to enter in the market (Berckmans et al., 2017; Lebedeva et al., 2016); as an example, the investigated chemistry LMO/NMC is last generation as compared to e.g. NMC 622 which is currently used in Chevrolet Bolt (with higher energy density). In general, the higher density of the next generation batteries will potentially result in higher lifetime and potential opportunities of reuse in different applications.

[83] For instance, it may be applied also to frequency regulation, especially considering that the TSO (Transmission System Operators) stakeholders shows especially high interest on frequency regulation ancillary services for relatively high potential revenues (Thien et al., 2017).

Figure 72: Schematic representation of the possible aspects that can be captured through the developed LCA model




Several stakeholders and the interest in facing legal barriers prove the relevance of economic aspects in extending the lifetime of batteries. Moreover, to have a complete overview of the sustainability of second-use of batteries, also social aspects should be assessed (BOX 9). In general, information to analyse both economic and social aspects are often missing or affected by uncertainty.
Moreover, the novelty of the topic, the available literature and the increasing number of research project underline the relevance of further work considering different aspects. Some examples are the following:

- how much of battery capacity available for repurposing can be absorbed by society?
- do multiple services at the same time make sense (i.e. EV providing other services such as Vehicle-to-grid)?
- What are the technical measures for improved business case (e.g. universal BMS, gathering info from first use to minimise effort for repurposing)?
- How can safety during transport of used batteries be improved (EV battery in car may be transported, but EV battery alone is considered hazardous good)?
- How could the analysis be enlarged to consider other applications (e.g. mobile second life charger; stationary battery off shore to quickly charge ships / storage in combination with fast charging installation) and batteries chemistries. Also, how could specific system characteristics (e.g. regulations related to electricity) be taken in account?
- How relevant aspects (e.g. technological development (e.g. solid state batteries, fast chargers); mobility patterns (car sharing, automated vehicles, etc.) and their consequences in EV batteries lifetime) could be better considered in the assessment? ;
- How could economic performances be assessed in order to identify the potential barriers/drivers for second-use of EV batteries.

# Material Flow Analysis. Analysis of the stocks and flows of traction batteries in Europe

The sustainability assessment of extending the lifetime of products can be captured through assessment tools considering a Life Cycle Thinking approach and multi-criteria analysis. However, available tools are not fully able to capture the impacts related to materials and resources (Ardente et al., 2017). Moreover, the penetration rate of a technology or the development of a business case related to reuse of products (in a large sense) depends of the availability of products in the market (Bobba et al., 2019; Rohr et al., 2017b).

The effects of extending the lifetime of products can be captured through the analysis of the stocks and flows of products (and related materials) along their value-chain (Mayer et al., 2018), which can also provide a more complete understanding of products' status (Nuss and Blengini, 2018). Therefore, the last methodological component of the proposed methodological framework is Material Flow Analysis, a methodology that aims to quantify the flows of materials and/or products along the main processes along the value chain, tracing the fate of the considered products/materials (Fischer-Kowalski et al., 2011; Nakamura et al., 2014). In particular, dynamic MFA allow to estimate the variation of stocks and flows along a period of time, reproducing historical stocks/flows and, according to the considered assumptions and scenarios, making projections for their possible development in the considered system boundary (Nakamura et al., 2014).

Especially in case of the potential development of new technologies, the knowledge of the variation of stocks and flow in Europe is an added value for both supporting policy decisions and for the stakeholders involved in the value chain of products. This is particularly relevant in case of strategic/critical products and materials for the EU economy. Based on the conclusions and follow up of the analysis on traction batteries, and the relevance of the topic (EC, 2018b; European Commission, 2018, 2017) the lifetime extension of traction Li-ion batteries have some potentialities in terms of environment impacts. However, the demand of resources needed to manufacture such batteries is rapidly increasing and among such resources there are also some Critical Raw Materials (CRMs) for Europe (COM(2017)490 final[84]), e.g., cobalt. Moreover, a new batteries' stock arises in the system, and the lifetime extension of LIBs results in an increased storage capacity in Europe that can potentially allow energy and greenhouse gases emissions savings (especially if batteries are coupled with renewable energy sources, e.g. PV panels). Meanwhile, recycling of embedded materials in LIBs will be postponed by a few years. All these aspects should be considered in order to have a more

[84] EC, 2017. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on the 2017 list of Critical Raw Materials for the EU. COM(2017)490 final

complete overview of the effects of second-use of traction batteries in Europe e to adopt a holistic approach for the analysis.

Accordingly, a Material Flow Analysis (MFA) was used to describe the stocks and flows of LIBs through the most relevant processes along the value chain of traction LIBs in Europe, from their use in xEVs until their recycling. Before describing the developed MFA model (section 5.2), the performed literature review is illustrated in section 5.1. Then, section 5.3 reports its application to the traction LIBs between 2005 and 2030 to estimate the variations of flows and stocks in the next decades according to the potential fast/slow development of second-use in Europe between 2005 and 2030. The same scenarios are adopted to also analyse the flows/stocks of both storage capacity and embedded materials. The selected materials for a first estimation are cobalt (as listed in the CRMs list of the EU) and lithium (as its demand is expected to fast increase due to the market demand).Finally, the main conclusions and follow-ups of the performed analysis are summarized in Section 2.3.

## 5.1 MFA of second-use of traction batteries

The performed literature review on MFA of second-use of batteries focused on the following aspects: temporal and geographical boundaries, types of battery, life-cycle steps reflected in the MFA, assessment of reuse in terms of both remanufacturing and second-use, and the criteria of the MFA analysis (e.g. products, materials). Table 31 summarises the main outcomes of the literature review in relation to the goal of this study.

The results of the broad-scope review confirmed that only a few studies provide a MFA that considers specific end-of-first-life options of LIBs after their removal from xEVs, including both remanufacturing and second-use. Among these, Busch et al. (2014) adopts highly speculative assumptions (95% of reuse of remanufactured batteries) to prove the potential of the proposed model. In other studies, the MFA estimates the flows potentially available for recycling/reuse without disaggregating the flows of recycling and/or reuse (Richa et al., 2014; Rohr et al., 2017b). Many authors do not consider the option of reuse at all.

The MFA studies analysed are mainly dynamic MFA performed up to 2030 (Rohr et al., 2017b), 2040 (Richa et al., 2014) and 2050 (Busch et al., 2014; Pehlken et al., 2017; Ziemann et al., 2018).

As highlighted by Ardente and Mathieux (2014), geographical and temporal representativeness is relevant for the assessment of various EoL scenarios, e.g. in relation to legislation in force or technological development of a specific area. However, 6 out of 10 studies developed the analysis at global level. Studies at national scale were performed by Busch et al. (2014) and Rohr et al. (2017b), who focused their MFA respectively on UK and Germany, whereas Richa et al. (2014) considers an intermediate scale (U.S.). Since the EU is "the second largest market of electric vehicles", the EU is the geographical boundary of the study performed by Simon et al. (2015); however, no detailed processes of the LIBs value-chain are described in the study. According to the authors' knowledge, no other dynamic MFA studies including second-use of LIBs has Europe as a geographical boundary. Because the EU level might be the right granularity to address the battery value-chain (cf. battery action plan for manufacturing step or Waste Battery Directive for end-of-life step), it is hence necessary to develop MFA studies for batteries at the EU level.

The relevance of assessing the demand for resources related to the fast increase of LIBs is recognised by the majority of the examined studies. However, data for this assessment are uncertain due to the scarcity of robust data (Olivetti et al., 2017) and the intrinsic level of uncertainty related to the development of new technologies as batteries (Majeau-Bettez et al., 2011; Pehlken et al., 2017). Even though roadmaps of LIBs are available in the literature, changes in LIBs technology and the increase/decrease of materials' content in the coming decades are considered only by Ziemann et al. (2018) (for Li). Other studies adopt a fixed materials breakdown of LIBs cells for various chemistries.

Finally, the energy storage capacity related to LIBs is usually assessed according to the installed capacity in the xEV market due to xEV demand. However, no details about the potential energy storage capacity of batteries after their use in xEVs is available. Despite the difficulties in estimating the batteries' lifetime and their residual capacity after they are removed from xEVs (Podias et al., 2018), the estimation of such capacity along the LIBs value-chain could offer a better understanding of the exploitable energy storage capacity.

Table 31: Summary of the most relevant aspects for this study available in the scientific literature

| Source | | Dynamic/Static MFA | Time frame | Scale of the study | | | Battery type | | "System Boundaries" of the analysis performed in the study | | | | Reuse/remanufacturing (i.e. is reuse/remanufacturing addressed in the study? If yes, how are they considered?) | Criteria of the analysis | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Global | Regional (region) | National (country) | LIB (type) | Others | Extraction | Manufacturing | Use | EoL | | Material | Product | Energy | Other |
| 1 | (Simon and Weil, 2013) | S (MEFA) | --- | --- | --- | --- | X (NMC, LFP) | --- | X | X | --- | X | --- | Al, Cu, steel, Li, Ni, Co, Mn | X (1 battery) | X | --- |
| 2 | (Busch et al., 2014) | D | 2010 - 2050 | --- | --- | X (UK) | X | --- | --- | --- | X | X | Maximum reuse rate is 95% (highly speculative) | Li, Co, Nd, Pl (fixed breakdown) | EV | --- | --- |
| 3 | (Nakamura et al., 2014) | D | 2005 for 100 y | --- | --- | X (Japan) | --- | --- | --- | X | X | X | --- | car steel | --- | --- | --- |
| 4 | (Richa et al., 2014) | D (future oriented MFA) | 2015 - 2040 | --- | X (U.S) | --- | X (NMC, LFP, LMO, LCO) | --- | --- | X | X | X | Recycling/Reuse not disaggregated | Al, Co, Cu, Ni, steel, iron, Li, Mn (fixed breakdown of the LIB) | X | --- | --- |

| | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 5 | (Reuter et al, 2014) | D | 2015 - 2050 | X | --- | --- | X (NMC, LFP) | --- | X | --- | --- | X | --- | Li, Ni, Mn, Co, iron, natural graphite, phosphate | --- | --- | --- |
| 6 | (Schmidt et al., 2016) | S | --- | X | --- | --- | X (NMC, NCA, LCO) | --- | X | X | X | --- | --- | Co, Ni | --- | --- | --- |
| 7 | (Pehlken et al., 2017) | D | 2014 - 2050 | X | --- | --- | X (NMC, LFP, LMO) | --- | --- | X | X | X | --- | Li, Co (fix breakdown ) | --- | --- | --- |
| 8 | (Rohr et al., 2017) | D | 2015 - 2030 | --- | --- | X (Germany) | X (NMC, LFP, NCA) | --- | --- | --- | --- | X | --- | Various materials assessed based on their price | X | --- | Price |
| 9 | (Sun et al., 2017) | D (dynamic trade-linked MFA) | 1994 - 2015 | X (trade flows) | --- | --- | X | Li in various products | X | X | (X) | --- | --- | Li | --- | --- | --- |
| 10 | (Olivetti et al., 2017) | forecasts, not a real MFA | 2002 - 2025 | X (trade flows) | --- | --- | X (NMC111, NMC622, NMC811, NCA, LCO) | --- | X | --- | --- | --- | Reuse discussed qualitatively | Co, Li, Mn, Ni, natural graphite | X | --- | --- |

| | | | | | | | | | | | | | | | | | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|---|
| 11 | (Ziemann et al., 2018) | D | 2010 - 2050 | X | --- | --- | X (NMC, NCA, Li-S) | --- | X | X | X | X | --- | Li (2010 and 2050) | --- | --- | --- |

NMC = lithium-nickel-manganese-cobalt cathode
NCA = lithium-nickel-cobalt-aluminium cathode
LFP = lithium-iron-phosphate cathode
LCO = lithium-cobalt-oxide cathode
LMO = lithium-manganese-oxide cathode
Li-S = lithium-sulphur cathode

## 5.2 The MFA model

The necessary background for the MFA mode is represented by the knowledge of the value chain of batteries in Europe. Since the understanding of the whole emerging system is still very limited and fragmented, a set of questionnaires was developed and adopted for managing the interviews to different stakeholders of the batteries' value chain. The questionnaires were based on literature information (e.g.(Canals Casals et al., 2015; Neubauer et al., 2015a)) as well as on the previous experience of JRC colleagues (Mathieux and Brissaud, 2010).

In order to gather information from all the actors of the chain, the following different questionnaires were realized:

- questionnaire for car companies;
- questionnaire for waste batteries collectors;
- questionnaire for re-purposing companies;
- questionnaire for actors using re-purposed batteries;
- questionnaire for experts

Then, the potential actors to interview have been identified and contacted. The questionnaires can be found in (Bobba et al., 2018b).

Table 32: List of the identified and contacted stakeholders

| | Actors | | Website | Feedback |
|---|---|---|---|---|
| 1 | Battery manufacturer | **EUROBAT** - Association of European Automotive and Industrial Battery Manufacturers | www.eurobat.org | Contacted - questionnaire sent |
| 2 | Car company | RENAULT | | Contacted - questionnaire sent |
| 3 | Car company | FCA | | Answered questionnaire |
| 4 | Car company | PEUGEOT/CITROEN | | Answered questionnaire |
| 5 | Car company | HYUNDAI MOTOR | | Answered questionnaire |
| 6 | Car company | MITSUBISHI | | Answered questionnaire |
| 7 | 2nd use project | **Bosch/BMW/Vattenvall** (pilot project: Second Life for electric-vehicle batteries) | http://www.bosch-presse.de/presseforum/details.htm?txtID=7067 | Answered questionnaire |
| 8 | Waste batteries collectors | **Battery Foundation** (Stichting Batterijen) in NL (Advised by Wecycle instead because they work in cooperation with ARN) | https://www.stibat.nl/ | Contacted |
| 9 | Waste batteries collectors | **ARN** - centre of expertise for sustainability and recycling in the mobility sector. | http://www.arn.nl/en/ | Answered questionnaire |
| 10 | Expert | **EGVIA** - European Green Vehicles Initiative Association | http://www.egvi.eu/about-egvia/organisation | Contacted - questionnaire sent |
| 11 | Expert | **IKERLAN** - Spanish knowledge transfer centre - Project Battery 2020 | http://www.ikerlan.es/en/ | Contacted |
| 12 | Expert | **VUB** - The Vrije Universiteit Brussel | http://www.vub.ac.be/en/ | Contacted - questionnaire sent |
| 13 | Expert | **ENEA** - National Agency for New Technologies, Energy and Sustainable Economic Development | http://www.enea.it/it | Contacted - questionnaire sent |

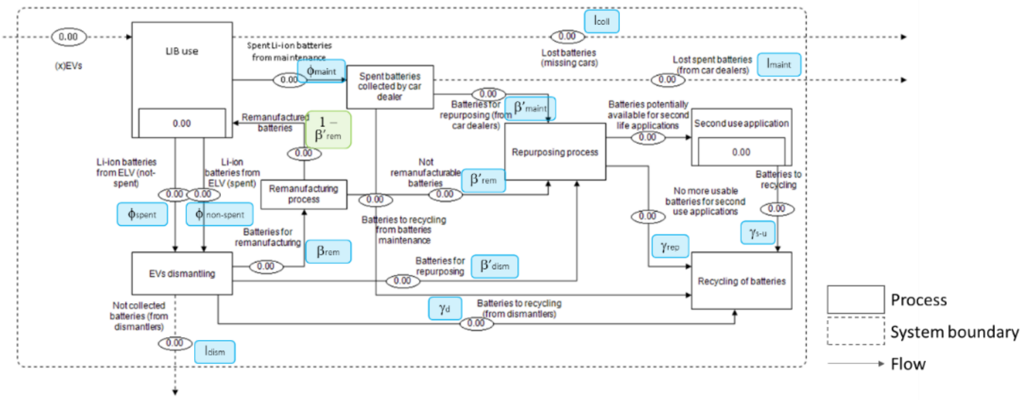
The detailed analysis of the answers is reported in (Bobba et al., 2018b). The outcome of the interviews, together with information available in the literature, allowed identifying the most relevant processes along the value chain of batteries in Europe, from their use to their disposal (Figure 73).

The life-cycle steps (represented as boxes in Figure 73) of the LIBs value-chain in Europe (Figure 73 - dashed box) were identified based on the stakeholders' interviews during the research. More information about interviews and interviewed stakeholders are available in Bobba et al. (2018b). Literature was used to complement when necessary. Flows of batteries between different processes are represented by arrows.

Traction batteries enter the European market through xEV sales; the first process of the system is therefore their use in xEVs ('LIB use'), where they are stocked according to their lifetime. After their use, batteries are collected through car dealers (e.g. due to recalls, malfunctions or accidents) ($\phi_{maint}$) or dismantlers (when no longer suitable for xEVs) ($\phi_{spent}$ and $\phi_{non\text{-}spent}$) (Timmers, 2016). In the model, batteries from end-of-life vehicles are handled by dismantlers ('EVs dismantling'), whereas spent batteries substituted during xEV maintenance are handled by car dealers ('Spent batteries collected by car dealers'). Since not all the batteries are properly removed and collected (Oko Institute, 2016), output flows represent the potential exports from the missing flow of xEV batteries ($l_{coll}$). Other potential losses along the LIBs value-chain are encompassed in the model through $l_{dism}$ and $l_{maint}$.

In Europe, exhausted batteries have to be recycled (EU, 2006, 2000). However, batteries can be removed due to their warranty conditions but still be usable in xEVs (Neubauer et al., 2015b; Willson, 2018). Therefore, before recycling they can be remanufactured and used again in a xEV: in this case, the battery is tested and reconditioned (if necessary) in Europe, and reused again in a xEV (APRA Europe, 2012). This option is considered through the 'Remanufacturing process' ($\beta_{rem}$). According to APRA, remanufacturing of batteries is currently developed in Europe.

Figure 73: Value-chain model of xEV batteries in Europe according to the interviewed stakeholders and the literature review




After being removed from xEVs, the residual capacity of batteries could potentially be exploited in other applications than xEVs ($\beta'_{rem}$, $\beta'_{dism}$, $\beta'_{maint}$). In the case of second-use, batteries are tested, repurposed (if needed) ('Repurposing process') and then used in different applications ('Second-use application'). This means that a new stock of LIBs within the system should be considered, and the 'Recycling of batteries' be delayed in line with the lifetime of LIBs in second-use applications (Rohr et al., 2017b). In line with the goal of the study, and since the landfilling of batteries is banned in Europe (EU, 2006), all batteries in the model are addressed to recycling (either after their first or second life).

## 5.2.1 Definition of the scenarios for the modelling of flows

The analysis aims to assess the effects of the potential second-use of xEV batteries in Europe, and, in order to test the responsiveness of the model, three different scenarios are considered.

The "Recycling" scenario ('REP-0') assumes that, after their removal from xEVs, batteries are collected and addressed to recycling. Considering the current situation in Europe, the market for both remanufacturing and second-use of batteries is not yet developed (section 1). Therefore, with a view to establishing a term of reference scenario, no remanufacturing and no second-use of EV batteries are considered in the 'REP-0 scenario'.

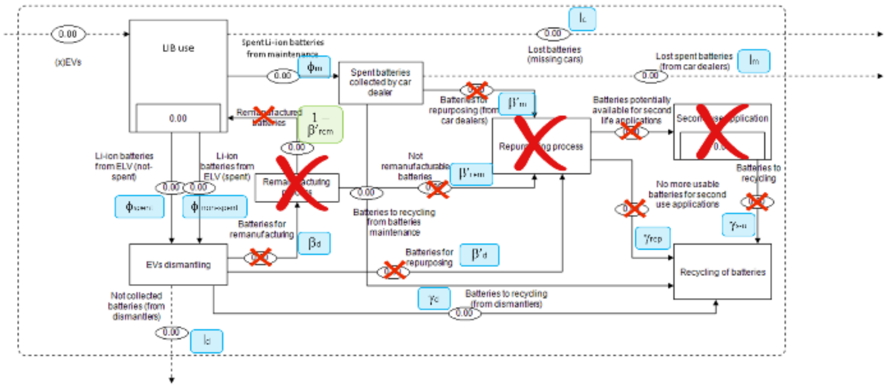
The 'REP-0 scenario' is the reference scenario for the comparison with other two scenarios that capture the potential development of a European market for second-use of xEV batteries: "Low second-use scenario" ('REP-20') and "High second-use scenario" ('REP-70'). Bearing in mind the existing barriers/drivers to the development of second-use of xEV batteries (e.g. incentives, legal framework, quantities of LIBs) (Elkind, 2014; Neubauer et al., 2015a; Reinhardt et al., 2017) but also the demonstrated interest in tackling these barriers (e.g. through the Innovation Deal of reuse of xEV batteries[1]), second-use of batteries could gradually develop in the near future. However, due to the novelty of this EoL strategy, the trend of second-use development is unknown and uncertain. In the case of second-using batteries, the batteries' lifetime is extended in line with both the battery and the application characteristics (Bobba et al., 2018a) and recycling of the battery is consequently postponed in time. The 'REP-20 scenario' captures the gradual development of LIBs second-use 20% in 2030 (Table 2) through an annual increase of batteries addressed to second-use. Moreover, in line with the current market, and in order to observe the variation of LIB flows related to the arising of second-use, no remanufacturing in Europe is considered in such a scenario.

Finally, the 'REP-70 scenario' was modelled to capture a potential fast development of reuse of batteries, through either remanufacturing or second-use. In their modelling scenarios, Neubauer et al. (2015a) approximate that 80%-90% of batteries will be eligible for repurposing, meaning that "significant deployments of second use batteries" will occur after 2030. Natkunarajah et al. (2015) consider that all the batteries could be adopted in second-use applications. Also, the model used by Standridge et al. (2016) envisages that 85% of the batteries will be usable in post-vehicle-applications. In conclusion, based on the literature and also in line with the goal of the paper, the 'REP-70 scenario' considers that 20% of the non-spent LIBs will be remanufactured (i.e. used again in xEVs), and the majority of the removed LIBs will be adopted in various applications other than in xEVs.

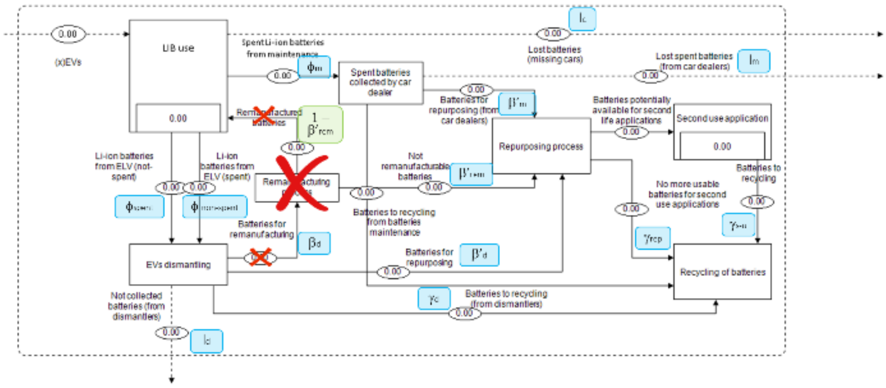
Figure 74 gives an overview of the main differences between flows assessed in the three scenarios illustrated above.

Figure 74: Differences in the value-chain processes in Europe in line with the assessed scenario. Red crosses highlight processes with no flows of batteries

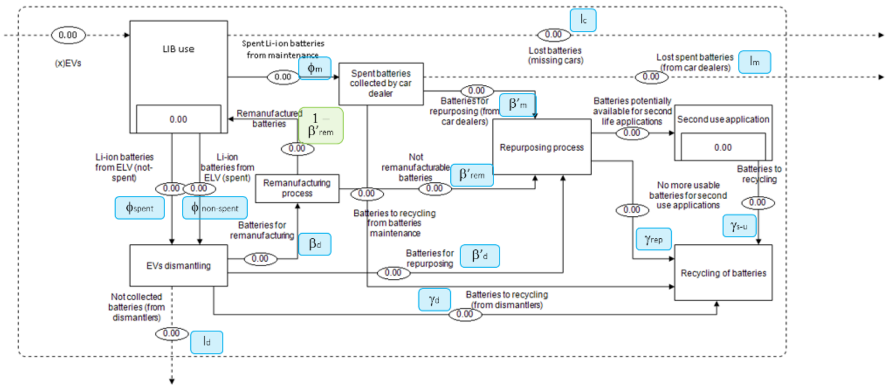
REP-0 scenario




REP-20 scenario




REP-70 scenario




## 5.2.2 Assessed aspects addressed by the stocks and flows model

The proposed model allows the assessment of the variation of xEV battery flows along the various processes of the value-chain in line with the input data. Furthermore, it is constructed in such a way that it enables to consider different aspects related to traction batteries, such as materials embedded in LIBs and/or energy capacity.

Throughout their lifetime, LIBs provide energy to xEVs but potentially also to other applications. The fact that a battery's capacity decreases during its lifetime depending on the battery's characteristics and use is one of the most relevant parameter to be considered for second-use applications (Podias et al., 2018; Rohr et al., 2017a). Considering the capacity of different types of batteries, the model can be used to estimate the flows of energy storage capacity associated with a battery's flows at different steps of its life.

Finally, the model can also be used to assess the stocks and flows of specific materials embedded in LIBs along their value-chain. Despite the intrinsic uncertainty related to new technologies (Pehlken et al., 2017), the model enables to estimate the flows of materials relevant for Europe, for instance cobalt or lithium embedded in specific LIBs chemistries.

## 5.3 The flows and stocks of LIBs in Europe

The MFA model is applied to the xEVs LIBs in Europe between 2005 and 2030, in particular to those used for both plug-in and full xEVs (i.e. PHEVs and BEVs). Hybrid electric vehicle (HEVs) LIBs were excluded mainly due to their characteristics: in HEVs, the conventional combustion engine is the main power source (electricity is generated on board) (EUROBAT, 2014; Huss et al., 2013; McEachern, 2012) and the level of electrification of HEV batteries is lower than for traction batteries used in BEVs and PHEVs. Consequently, also according with (EUROBAT, 2015), second-use of LIBs is considered only for PHEVs and BEVs.

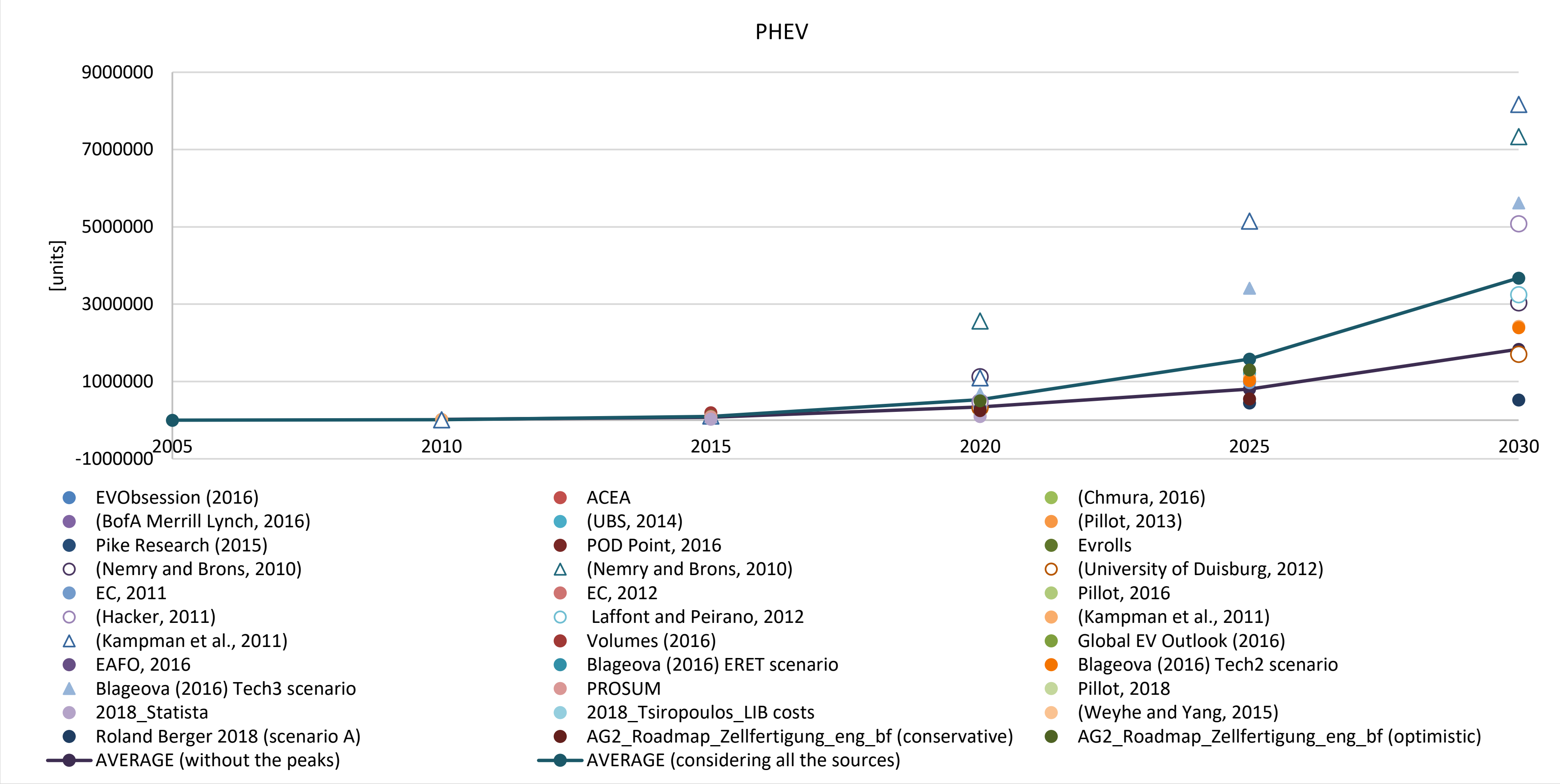
### 5.3.1 Data and assumptions to model the stocks and flows of traction LIBs

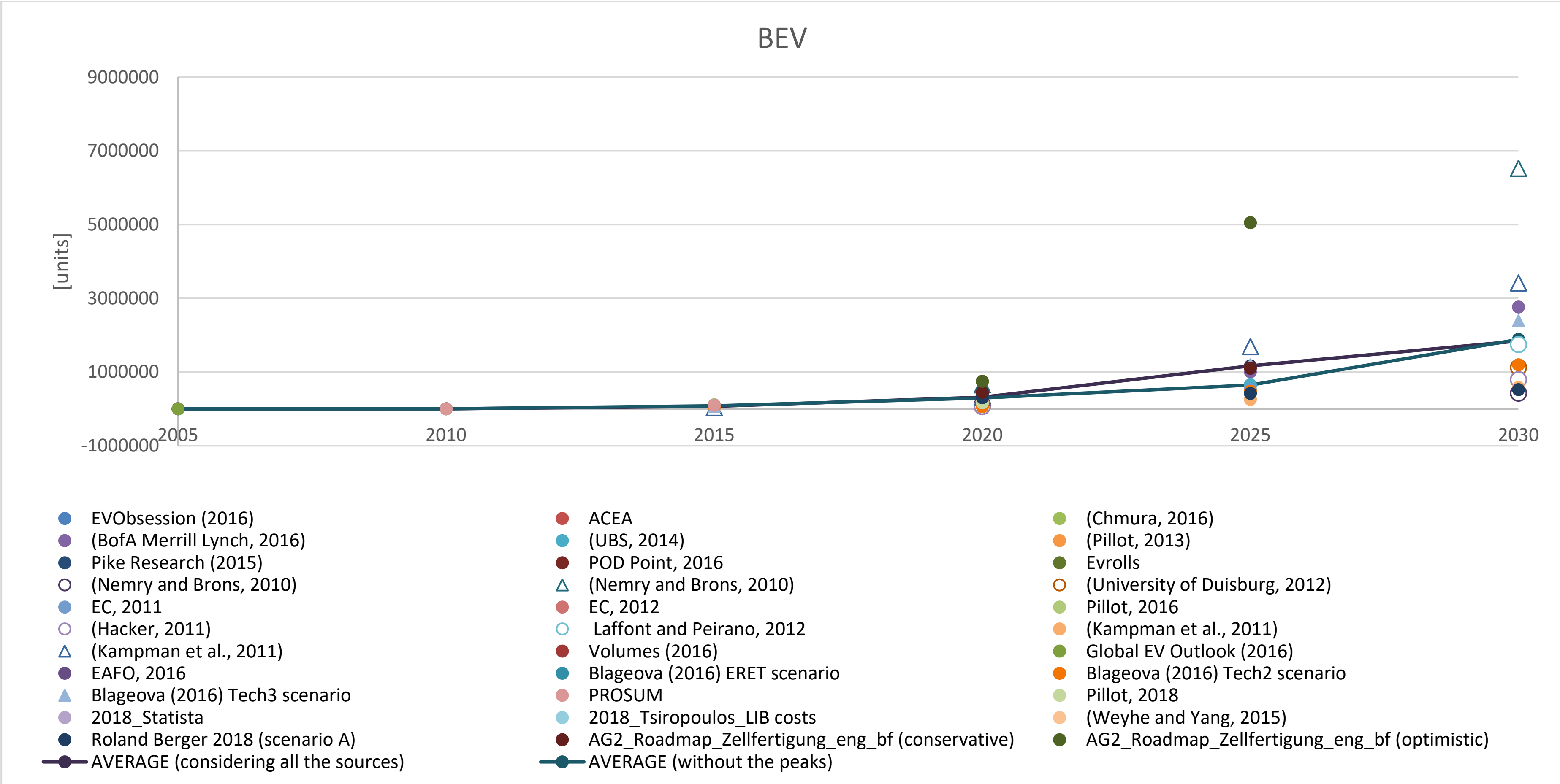
Consistent with the MFA methodology, the law of conservation of matter is used to establish the metric calculation and the relationships between the processes of the system (Brunner and Rechberger, 2004; Müller et al., 2014). STAN software[2] is used to estimate the stocks and flows of the system for all the assessed scenarios.

**xEV sales**

The estimate of BEVs and PHEVs sales in Europe between 2005 and 2030 is based on several sources available from the literature, e.g. Bank of America Merrill Lynch (2016); Blagoeva et al. (2016); EAFO (2016); Kampman et al. (2011). Own calculations were necessary since often data are aggregated or provided at global level. Excluding peaks of sales mainly related to optimistic scenarios (e.g. in Kampman et al. (2011)), the elaboration of projected European sales confirmed the trend illustrated by Lebedeva et al. (2016) and (Alves Dias et al., 2018) (Figure 75). In the model, the time interval considered for the analysis is 1 year.

Figure 75: Projected sales of new PHEV and BEV vehicles in Europe between 2005 and 2030 according to different sources. Empty markers refer to sources before 2015. The line correspond to average data (with and without the peaks)




BEV
[units]
9000000
7000000
5000000
3000000
1000000
-1000000
2005
2010
2015
2020
2025
2030
EVObsession (2016)
(BofA Merrill Lynch, 2016)
Pike Research (2015)
(Nemry and Brons, 2010)
EC, 2011
(Hacker, 2011)
(Kampman et al., 2011)
EAFO, 2016
Blageova (2016) Tech3 scenario
2018_Statista
Roland Berger 2018 (scenario A)
AVERAGE (considering all the sources)
ACEA
(UBS, 2014)
POD Point, 2016
(Nemry and Brons, 2010)
EC, 2012
Laffont and Peirano, 2012
Volumes (2016)
Blageova (2016) ERET scenario
PROSUM
2018_Tsiropoulos_LIB costs
AG2_Roadmap_Zellfertigung_eng_bf (conservative)
AVERAGE (without the peaks)
(Chmura, 2016)
(Pillot, 2013)
Evrolls
(University of Duisburg, 2012)
Pillot, 2016
(Kampman et al., 2011)
Global EV Outlook (2016)
Blageova (2016) Tech2 scenario
Pillot, 2018
(Weyhe and Yang, 2015)
AG2_Roadmap_Zellfertigung_eng_bf (optimistic)

## Penetration rate

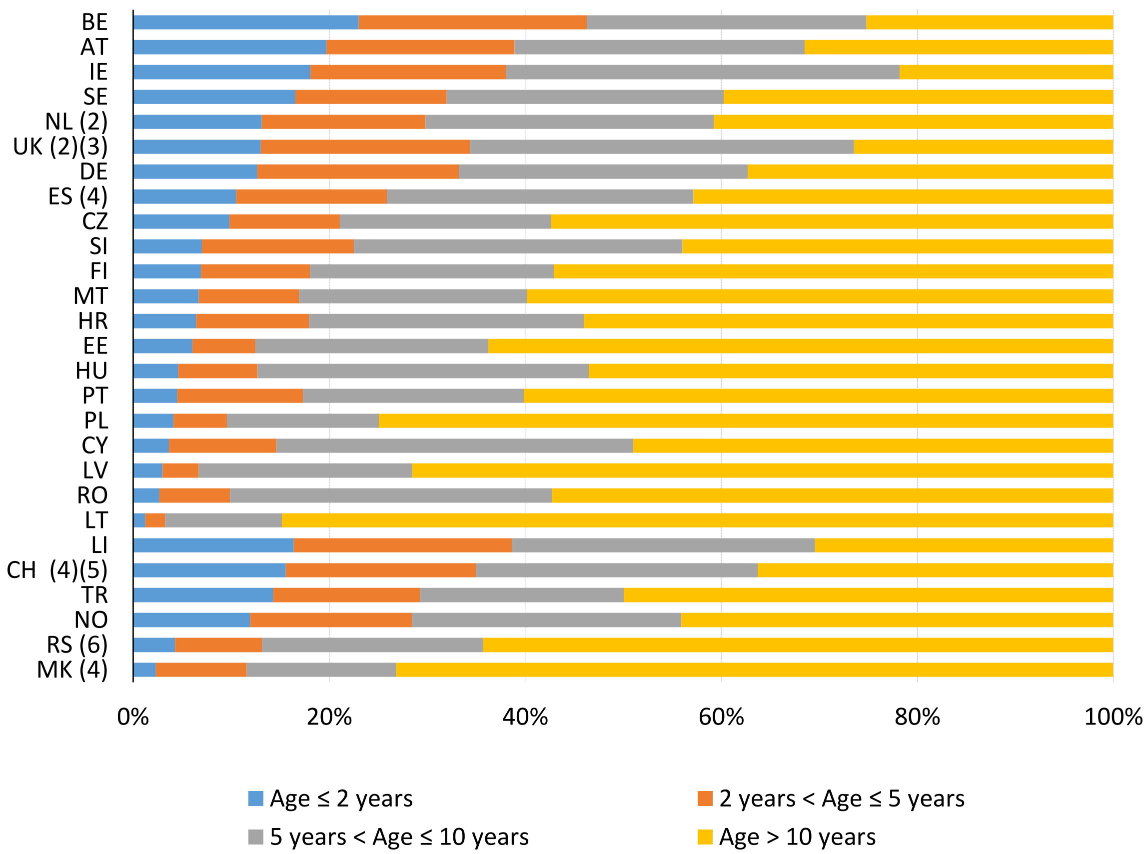
A EUROBAT analysis (EUROBAT, 2014) confirmed that PHEV propulsion batteries are only Li-ion batteries, for full HEVs both Li-ion and Ni based batteries are used and Li-ion batteries predominate the BEVs (Chmura, 2016; Gasparin, 2015). The competitiveness growth will be expected in battery cost reduction, energy and power density increase, longer lifetime and increased charge acceptance.

For the analysis, a penetration rate for LIB in xEVs of 70%, 80% and 100% (linear increasing) is assumed respectively for 2005, 2010 and after 2015.

## Average lifetime

EUROSTAT data show that 43.90 % of passenger cars in 2013 had a lifetime higher than 10 years, whereas the 27.37 %, lower than 5 years (EUROSTAT, 2018). It is worthy that various aspects contribute to make this aspect highly uncertain and this value varies across different studies (Richa et al., 2014; Sweeting and Winfield, 2012).

Figure 76: Cars lifetime per European Country (EUROSTAT, 2015)




Battery lifetime depends on several factors, e.g. driving style and frequency of charging (Daimler, 2015; Podias et al., 2018). In the international literature, the average lifetime of Li-ion batteries ranges between 5 and 15 years (Ahmadi et al., 2014a; Canals Casals et al., 2015; Neubauer et al., 2015a; Richa et al., 2015, 2014; Sathre et al., 2015). The batteries requirements for HEVs, PHEVs and EVs illustrated by EUROBAT (2014) stated that the service life expectancy is at least 10 years, even though the EUROSTAT E-mobility Roadmap points out that EV cells are expected to achieve 15 years in 2030 (EUROBAT, 2015). This is aligned with manufacturers' warranties, e.g. Leaf battery[85] (8 years/100,000 miles) (Cobb, 2014). A survey carried out by Tesla (section 3.3) state that most of the sample batteries loos 10% of their capacity after 270,000 km. In general, consumers expect that the lifetime of traction batteries is at least long as the lifetime of the vehicle (Circusol Workshop (11/03/2019), BOX 2).

In general, traction LIBs are removed from xEVs for different reasons, e.g. low capacity (spent battery), end of the warranty/leasing period, accidents.

In case of lack of data about lifetime of products, mathematical models can be used, e.g. Weibull distributions (Mathieux and Brissaud, 2010; Müller et al., 2014). For the performed analysis, in order to check the potentiality of the model and to capture this variability, a discrete distribution of the batteries lifetime is assumed according to Richa et al. (2014). The discrete distribution captures e.g. possible early replacements, EoL users' behaviours considering that 10% of batteries placed on market has a lifetime of 6 years, 40% of batteries has a lifetime of 8 years and the reaming 10% of batteries has a lifetime exceeding 12 years.

The uncertainty of such aspects would require a more in-depth analysis and real data to estimate the real lifetime of LIBs in both first and second life (Podias et al., 2018).

**Collection rate**

The first step of the potential reuse of xEV batteries is the collection after their removal from xEV and their sorting. As declared by EC (2014c), the collection rate of both automotive and industrial batteries in Europe is nearly 100%. Therefore, a high availability of xEV batteries after their use in xEV is expected also in the future.

It is underlined that the expected increase of the xEV market results in an increasing waste batteries flow to be managed. A "reverse logistics"[86] effort could optimize the retailer supply chain and minimize the operational and environmental costs (Klör et al., 2014; Pourmohammadi et al., 2008; Roghanian and Pazhoheshfar, 2014; Schultmann et al., 2003), strength the system effectiveness and decrease costs (CEC, 2015; Groen, 2016).

Concerning Li-ion xEV batteries, an appropriate and safe removal, handling and transport of such batteries is needed (Van Peperzeel communication) and could minimize the failure rate of repurposing operations (Ahmadi et al., 2014b; Canals Casals and Amante García, 2016; CEC, 2015). Then, both specialization of operators who can safely manage batteries (Groen, 2016) and strengthening of stakeholders network (CEC, 2015; IHS Consulting, 2014) are two relevant aspects for potentially ease the second-use of xEV batteries.

(Ruiz et al., 2016) identified car manufacturers as key players in this process due to their access

---

[85] http://www.hybridcars.com/how-long-will-an-evs-battery-last/, https://steinbuch.wordpress.com/2015/01/24/tesla-model-s-battery-degradation-data/ (survey dated 2015)

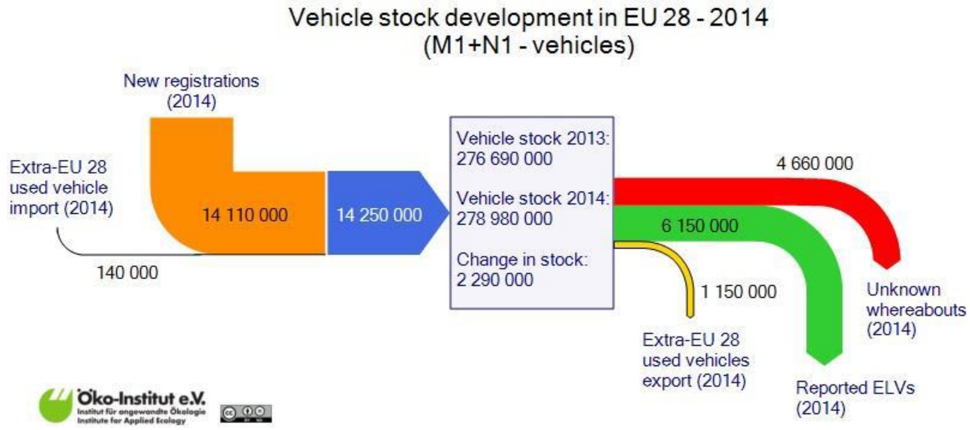
[86] The reverse logistics is defined as "the process of planning, implementing, and controlling the efficient, cost effective flow of raw materials, in-process inventory, finished goods and related information from the point of consumption to the point of origin for the purpose of recapturing value or proper disposal" (American Reverse Logistics Executive Council)

to technical information and their interest in the topic as they might be owner of the battery pack and obtain economic advantages from the batteries reuse.

Until 2016, the minimum collection rate established by the Directive 2006/66/EC was 45%. However, there are already examples of very high collection rate of both automotive and industrial batteries in Europe (e.g. 91% for Toyota and Lexus)[87], and some studies asserted that the collection rate of the automotive and industrial batteries is expected to be very high (EC, 2014c; Mudgal et al., 2014).

Nonetheless, about 40 % of the vehicle waste flow (including batteries) in the EU is unknown whereabouts (Oeko Institute, 2017) and, according to the consulted stakeholders, the abovementioned collection rate is overestimated.

Figure 77: Vehicle park development in Europe – 2013 (Oeko Institute, 2017)




Due to the lack of data on collection of traction LIBs (Stahl et al., 2018), an initially conservative collection rate is assumed for 2005 (60%) and then it is assumed to rise constantly to 90% in 2030.
It is also assumed that batteries collected by car dealers have already reached their EoL, so that they are no longer usable in xEVs; in this case, the analysis entails the substitution of the battery only if the car still has more than 2 years' lifetime.

Based on these considerations, the available EV batteries in the waste stream in a specific year ($Wb_t$) depends on both the lifetime of the EV ($l_{EV}$) and the lifetime of the batteries ($l_b$):

$$WSb_t = \sum(Sb_{t-l_b} + Wb_{t-l_{EV}} + Rb_{t-l_{EV}+l_b})$$

Where:

- $WSb_t$ = available EV batteries in the waste stream in a specific year ;
- $\mathrm{Sb}_{t-l_b}$= EV batteries which have reached their EoL ;
- $\mathrm{Wb}_{t-l_{EV}}$ = EV batteries withdrawn due to the EV EoL;
- $\mathrm{Rb}_{t-l_{EV}+l_b}$ = EV batteries replaced due to the previous battery EoL.

---

[87] http://www.autoblog.com/2015/02/10/toyota-mirai-most-innovative-honor/

**Repurposing and second-use**

Batteries potentially adoptable in second-use applications should be tested to assess their conditions (e.g. state-of-health) and the best suitable application (Koch-Ciobotaru et al., 2015; Rehme et al., 2016). Defining the lifetime of batteries in such applications is challenging since it depends on both the battery's and the systems' characteristics; also, a lack of data is often addressed through estimates or average data (Bobba et al., 2018a, 2018b). Based on an average value of 8 years, this aspect was varied in the sensitivity analysis in order to assess its relevance to the overall results.

Table 33 summarises the main assumptions and the main differences between the three scenarios illustrated above.

Table 33: Assumptions for the assessed scenarios

| **Flow/Process** | **Parameter** | **REP-0 SCENARIO** | **REP-20 SCENARIO** | **REP-70 SCENARIO** |
|---|---|---|---|---|
| Lost batteries (missing cars) | $l_{coll}$ | Annual linear decrease from 40% (in 2005) to 10% (in 2030) | Annual linear decrease from 40% (in 2005) to 10% (in 2030) | Annual linear decrease from 40% (in 2005) to 10% (in 2030) |
| Remanufacturing | $\beta_{rem}$ | 0% | 0% | 20% |
| Batteries for repurposing (from dismantlers) | $\beta'_{dism}$ | 0% | Annual linear increase from 0% (in 2005) to 20% (in 2030) | 70% |
| Not collected batteries(from dismantlers) | $l_{dism}$ | 10% | 10% | 10% |
| Batteries to recycling (from dismantlers) | $\gamma_{dism}$ | 100% - ($\beta_{rem}$ - $\beta'_{dism}$ - $l_{dism}$) | 100% - ($\beta_{rem}$ - $\beta'_{dism}$ - $l_{dism}$) | 100% - ($\beta_{rem}$ - $\beta'_{dism}$ - $l_{dism}$) |
| Not remanufacturable batteries | $\beta'_{rem}$ | 0% | 0% | 20% |
| Batteries for repurposing (from car dealers) | $\beta'_{maint}$ | 0% | Annual linear increase from 0% to 20% | 100% |
| Lost spent batteries (from car dealers) | $l_{maint}$ | 0% | 0% | 0% |
| Batteries to recycling (from car dealers) | $\gamma_{maint}$ | 100% - ($\beta'_{maint}$ - $l_{maint}$) | 100% - ($\beta'_{maint}$ - $l_{maint}$) | 100% - ($\beta'_{maint}$ - $l_{maint}$) |
| Batteries to recycling (from second-use applications) | $\gamma_{s\text{-}u}$ | INPUT | INPUT | INPUT |
| No more usable batteries for second-use applications | $\gamma_{rep}$ | 0% from $\beta'_{maint}$<br>0% from $\beta'_{rem}$<br>0% from $\beta'_{dism}$ | 0% from $\beta'_{maint}$<br>0% from $\beta'_{rem}$<br>10% from $\beta'_{dism}$ | 0% from $\beta'_{maint}$<br>0% from $\beta'_{rem}$<br>10% from $\beta'_{dism}$ |

### 5.3.1.1 Data and assumptions used to model the stocks and flows of energy storage capacity

Battery capacity (and its consequent lifetime) is an important limiting factor for the development of xEVs, and continuous efforts by the automotive and batteries industries are tending to increase it (Bank of America Merrill Lynch, 2016; EEA, 2016; Ziemann et al., 2018; Zubi et al., 2018). Due to confidentiality issues, few data about the forecasted capacity of traction batteries are available in the literature. In contrast to Simon and Weil (2013), in which fixed values are adopted to assess the flows of energy storage capacity in the near future, several sources were used to estimate the evolution of LIBs capacity over time (Table 35).

It is assumed that the capacity of LIBs when reaching their EoL (i.e. '$\phi_m$' and '$\phi_{spent}$') is 60% of the nominal capacity of the battery, whereas for other batteries (i.e. '$\phi_{non\text{-}spent}$') it is 80%.

#### 5.3.1.2 Data and assumptions used to model the stocks and flows of embedded materials

Li-Co based chemistries will remain the most promising chemistry for e-mobility before 2020 (Zubi et al., 2018). As a consequence of their increasing demand, demand of both Li and Co is also expected to increase substantially (IEA, 2018; Langkau and Tercero Espinoza, 2018).

Among the Li-Co chemistries, the NMC (nickel-manganese-cobalt) and NCA (nickel-cobalt-aluminium) are the most widely adopted for BEVs and PHEVs due to their suitable characteristics (e.g. energy density and durability) and the forecasted decrease of costs (Zubi et al., 2018). Then the model was applied to Co and Li embedded in these two chemistries. Due to the lack of data about the market share of such chemistry up to 2030, several sources were used to gather information (Bank of America Merrill Lynch, 2016; Berman et al., 2018; Blagoeva et al., 2016; Donat Marques, 2018; IEA, 2018; JRC, 2013; Mirae Asset Daewoo Co. Ltd., 2017; Pehlken et al., 2017; Pillot, 2017; Research, 2016). Results of elaborations and average shares are summarised in Table 34.

Concerning materials content, also because the cost of Co supply heavily affects the price of battery packs, its proportion in LIBs is expected to decrease after 2025 in different chemistries. For instance, new chemistries with lower Co content are available already, e.g. NMC 523, 622, and 811 instead of NMC 111 (Berman et al., 2018; IEA, 2018; Perks, 2016; Pillot, 2017) also, the use of composite cathodes is another strategy to decrease the Co content (Cusenza et al., 2019; Patry et al., 2014). Mainly due to the lack of data and the uncertainty of sources, steady values concerning materials content are usually used to assess the materials flow. However, to quantify the flows of specific materials along the various processes of the value-chain, technology development should be considered. Also in this case, several sources were consulted (Blagoeva et al., 2016; Gruber et al., 2011; JRC, 2013; Petersen, 2018; Tivander, 2016; Ziemann et al., 2018). Note that in 2030 all the LIB market is assumed to be made of NMC and NCA chemistries

Table 35 depicts the analysis inputs.

Table 34: Market share of NMC and NCA batteries included in the analysis.

| | NMC 111 | NMC 532 | NMC 622 | NMC 811 | TOT NMC | NCA |
|---|---|---|---|---|---|---|
| 2005 | 30.00% | 0.00% | 0.00% | 0.00% | 30.00% | 8.00% |
| 2010 | 12.00% | 18.00% | 0.00% | 0.00% | 30.00% | 10.00% |
| 2015 | 12.04% | 16.17% | 5.16% | 1.03% | 34.41% | 11.55% |
| 2020 | 23.95% | 20.96% | 11.97% | 2.99% | 59.87% | 10.97% |
| 2025 | 15.33% | 21.46% | 18.40% | 6.13% | 61.32% | 9.90% |
| 2030 | 9.00% | 27.00% | 36.00% | 18.00% | 90.00% | 10.00% |

Note that in 2030 all the LIB market is assumed to be made of NMC and NCA chemistries

Table 35: Summary of the data used for both the energy flows and the material content flows analysis

| | | Residual capacity [kWh/battery] | Cobalt content [kg/battery] | | Lithium content [kg/battery] | |
|---|---|---|---|---|---|---|
| | | | NMC 111 | NCA | NMC | NCA |
| **PHEV** | 2005 | 6.23 | 2.38 | 1.44 | 0.79 | 1.25 |
| | 2010 | 6.23 | 3.38 | 1.55 | 0.79 | 1.25 |
| | 2015 | 8.10 | 4.38 | 1.65 | 2.01 | 2.03 |
| | 2020 | 10.11 | 5.75 | 2.50 | 2.49 | 2.09 |
| | 2025 | 11.23 | 6.56 | 2.88 | 3.19 | 2.67 |
| | 2030 | 12.98 | 6.56 | 2.88 | 3.88 | 3.26 |
| **BEV** | 2005 | 17.58 | 14.34 | 8.44 | 4.64 | 6.23 |
| | 2010 | 17.58 | 14.04 | 8.13 | 4.64 | 6.23 |
| | 2015 | 28.75 | 13.74 | 7.81 | 5.49 | 6.09 |
| | 2020 | 38.70 | 20.98 | 9.12 | 7.43 | 7.62 |
| | 2025 | 39.65 | 20.83 | 9.14 | 8.27 | 8.48 |
| | 2030 | 45.20 | 20.83 | 9.14 | 8.86 | 9.08 |

* for the calculations, the Co percentages in the cathode are: 18.24% for NMC532, 12.16% for NMC622 and 6.06% for NMC811.

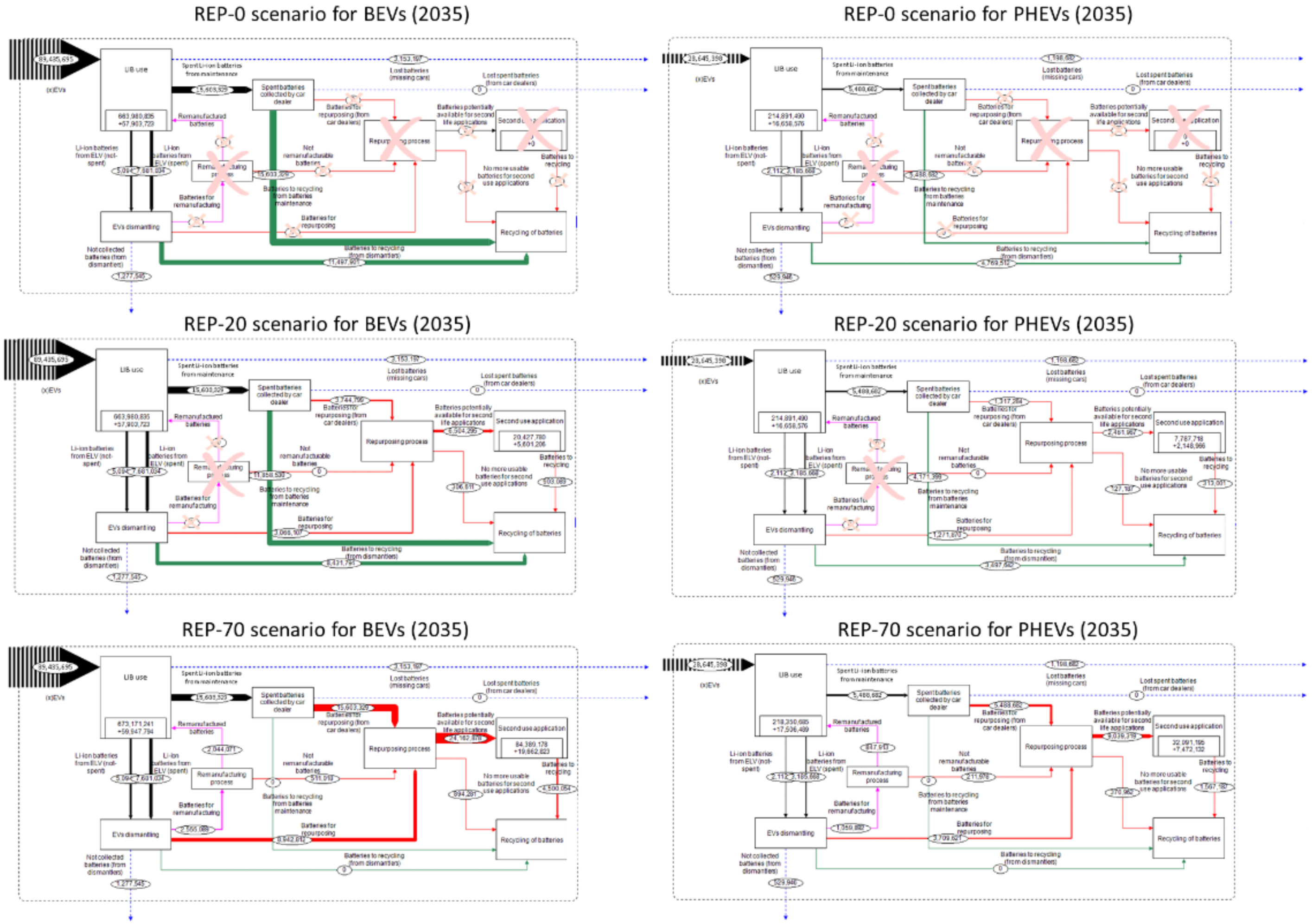
### 5.3.2 Results

Through the STAN software, the graphs illustrated in Figure 78 were obtained for the different assessed scenarios. Blue arrows represents the output flows (i.e. losses), red arrows correspond to the second-used LIBs flows, green arrows correspond to recycling flows of LIBs after their removal from xEVs, pink arrows correspond to the flow of remanufactured LIBs.

In general, the gradual increase of xEVs sales in Europe will not significantly affect the materials and capacity flows before 2025. Then, major differences will concern the recycling flows of LIBs after their removal from xEVs (green flows) and the second-used LIBs (red flows).

With a gradual increase of repurposing of xEVs batteries ('REP-20' scenario), in 2025 more than 38,500 LIBs could be adopted in second-use applications in Europe, of which 53% will be from PHEVs and 47% from BEVs (Figure 79). Results show that this amount of batteries corresponds to a residual capacity of 0.6 GWh: in turn this corresponds to about 14% of the energy storage capacity for self-consumption applications in Europe (Kessels et al., 2017). Even though the amount of BEVs and PHEVs batteries available for second-use is similar (about 70,500 and 91,500 respectively in 2030), about 73% of the abovementioned capacity is provided by batteries used in BEVs, which are characterised by higher energy density than batteries used in PHEVs. Focusing on the 'REP-70' scenario, the amount and the capacity of batteries available for second-use is 4 times higher in comparison to the 'REP-20' scenario (Figure 79).

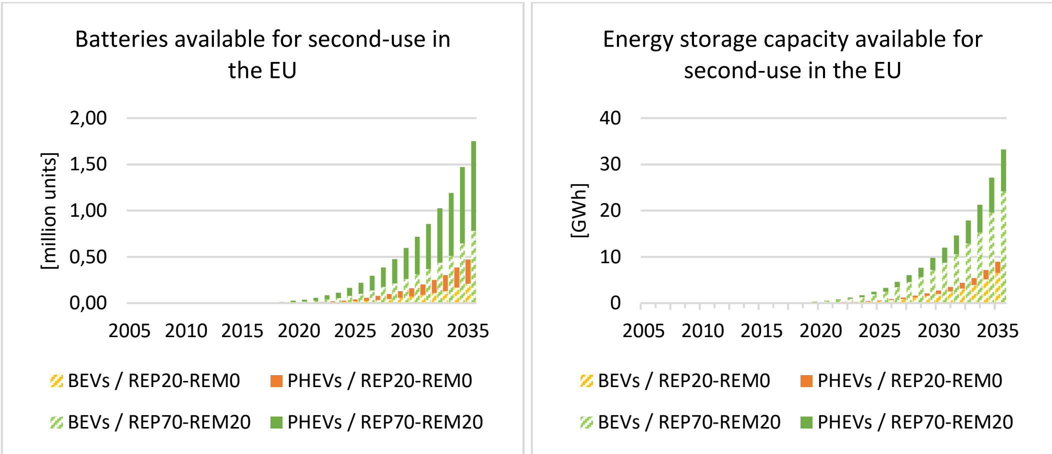
Figure 78: Energy capacity storage of LIBs in BEVs (left) and PHEVs (right) in 2035 in Europe for different scenarios



In the case of a gradual development of second-use, LIBs sent for recycling in 2030 and 2035 are estimated to be respectively 1.23 and 1.25 times lower than those in the 'REP-0' scenario, where no second-use occurs. In this case, through the model, it is possible to estimate the energy storage capacity of non-exploited LIBs due to direct recycling rather than second-use: in 2030, it is about 13.5 GWh for the 'REP-0' scenario, almost 11 GWh for the 'REP-20' scenario and almost 2 GWh for the 'REP-70' scenario (73% from BEVs' LIBs).

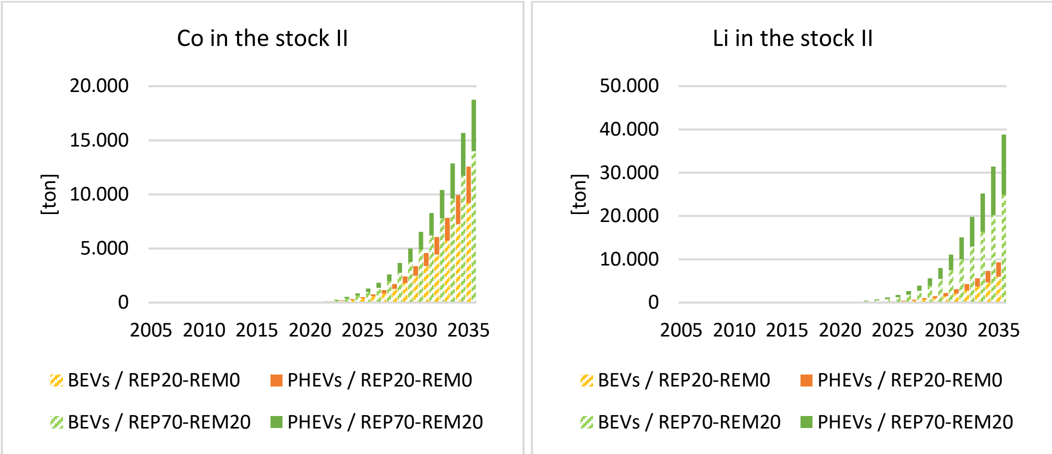
The delay in terms of available LIBs entering the recycling process can be estimated for the different scenarios: in 2020, about 40,500 LIBs are sent for recycling in the 'REP-0' scenario. The same amount will be recycled with a delay of half a year in the 'REP-20' scenario and of 7 years for the 'REP-70' scenario.

Figure 79: Batteries available for second-use applications in Europe (left) and the respective energy storage capacity (right).
The 'REP-0' scenario is not reported since no second-use occurs.



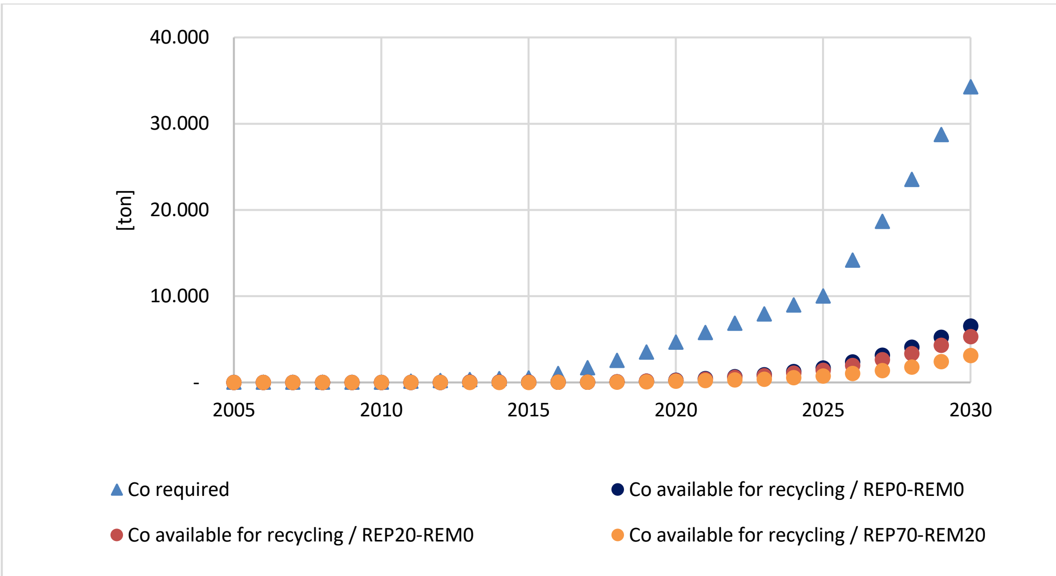

Second-use of LIBs results in the creation of new stock. Looking at materials embedded in LIBs, it is possible to estimate the amount of Co and Li stocked in second-use applications and consequently the time shift before they are sent for recycling. Focusing on the 'REP-20' scenario, in 2030 about 3,400 tonnes of Co will be stocked in xEV LIBs adopted in second-use applications (74% of which will be embedded in BEVs' LIBs). This amount is almost 2 times higher in the 'REP-70' scenario. This means that the Co available for recycling in 2030 is 19% lower in the 'REP-20' scenario than the 'REP-0' scenario. Similarly for Li, in 2030 the stock of Li in second-use batteries will be about 2,200 tonnes (67% of which in BEVs' LIBs) in the 'REP-20' scenario and about 11,000 tonnes in 'REP-70' scenarios (Figure 80).

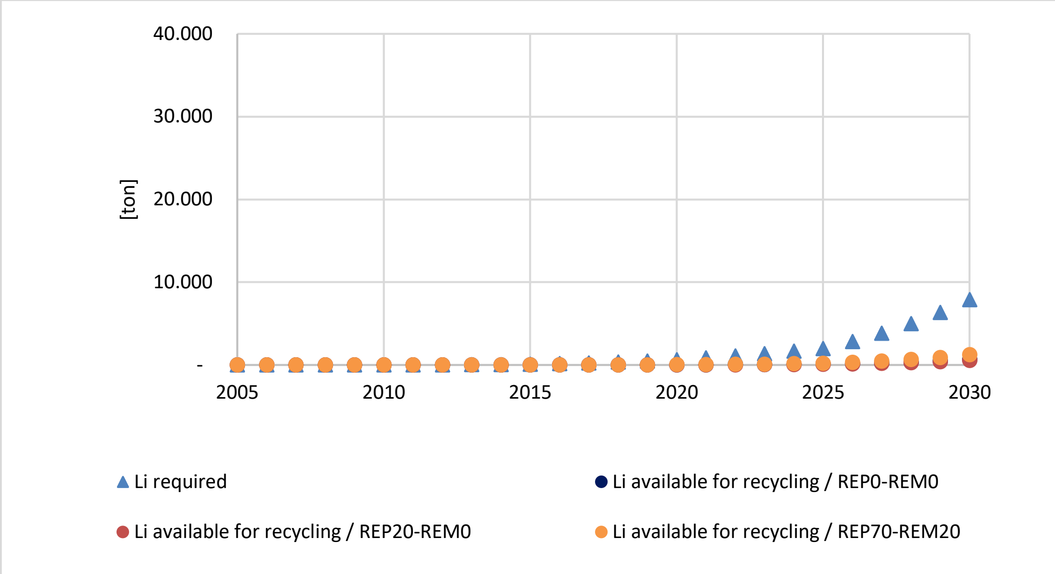
Figure 80: Cobalt and Lithium stocked in second-use applications in Europe.
The 'REP-0' scenario is not reported since no second-use occurs.




To have a more complete overview of the amount of materials entering the system through LIBs and the materials available for recycling, these flows are illustrated in Figure 81. Assuming that all the Co and Li could be used for LIB manufacturing, results show that the delay of Co and Li available for recycling caused by the second-use of LIBs does not significantly decrease the materials required for LIB manufacturing. For instance, in 2030 the Co entering the recycling process through LIBs ranges between 3,000 tonnes ('REP-70') and 6,500 tonnes ('REP-0') whereas the Co entering the EU embedded in LIBs is greater than 34,000 tonnes. Moreover, it is worth noting that the quantity of SRMs should include the efficiency of the recycling processes according to the technology applied. Currently, 94% of the input Co can be recovered, whereas Li recovery requires more complex treatment and its recovery is still not available in Europe at industrial scale (Lebedeva et al., 2016; Mathieux et al., 2017).

Figure 81: Cobalt and Lithium in LIBs and available for recycling in Europe






## Sensitivity analysis

Both the lifetime of LIBs in first and second life and LIB capacity are varied through a one-at-time variation (Igos et al., 2018).

Concerning the lifetime of LIBs in second-use applications, upper and lower values of lifetime are considered for the sensitivity analysis. According to the literature (Bobba et al., 2018a, 2018b), lower and upper values are respectively 5 and 12 years. Figure 82 and Figure 83 show the variation of the Co and Li stocked in second-use application when battery lifetime is varied, while Figure 84 and Figure 85 show the variation of the available Co and Li for recycling when battery lifetime is varied.

The higher variation refers to the amount of Co and Li stocked in second-use applications. For Co (Figure 82), the Co stocked in second-use application in 2030 in the longer lifetime (i.e. 12 years) is about 500 tonnes higher than in the shorter lifetime (i.e. 5 years). For Li (Figure 83), this difference is about 350 tonnes. Focusing on the available materials available for recycling, results show that the variation of the lifetime in second-use applications does not significantly affect the flows of materials entering in the recycling process. However, in terms of absolute values, the difference between the Co available for recycling (Figure 84) in the longer second-life is about 260 lower than in the shorter second-life. This difference is about 180 tonnes for Li (Figure 85).

Figure 82: Variation of the Co stocked in second-use applications in Europe with the variation of the lifetime of batteries during their second-use

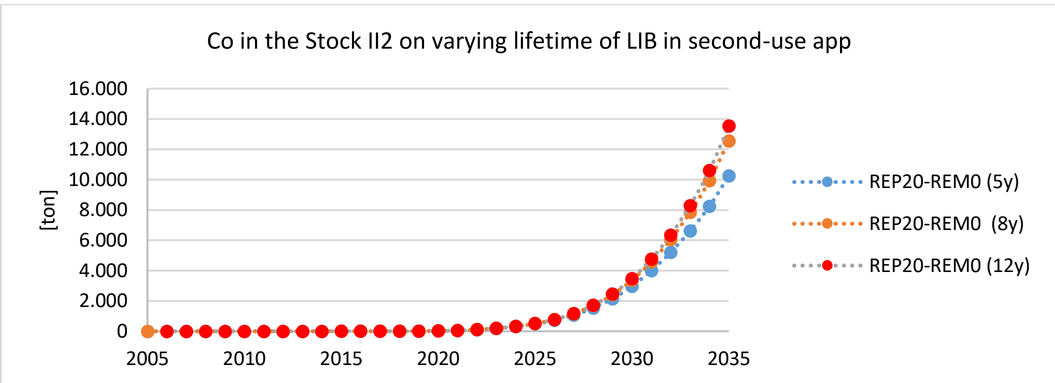


Figure 83: Variation of the Li stocked in second-use applications in Europe with the variation of the lifetime of batteries during their second-use

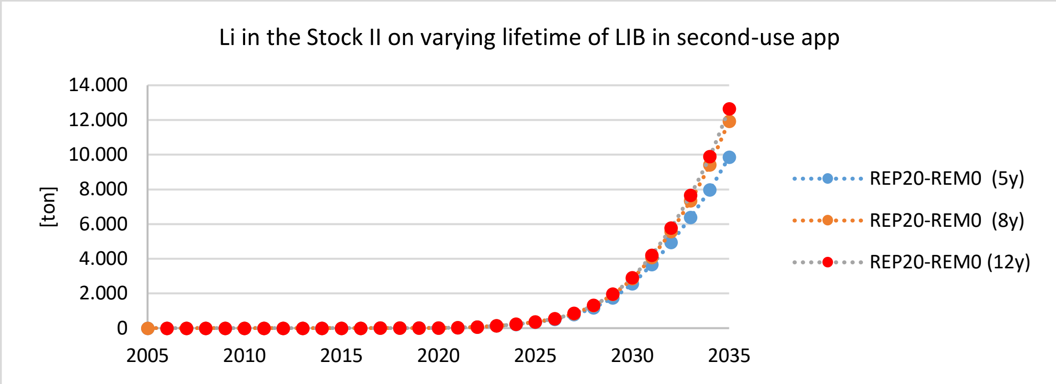



Figure 84: Variation of the Co available for recycling with the variation of the lifetime batteries during their second-use

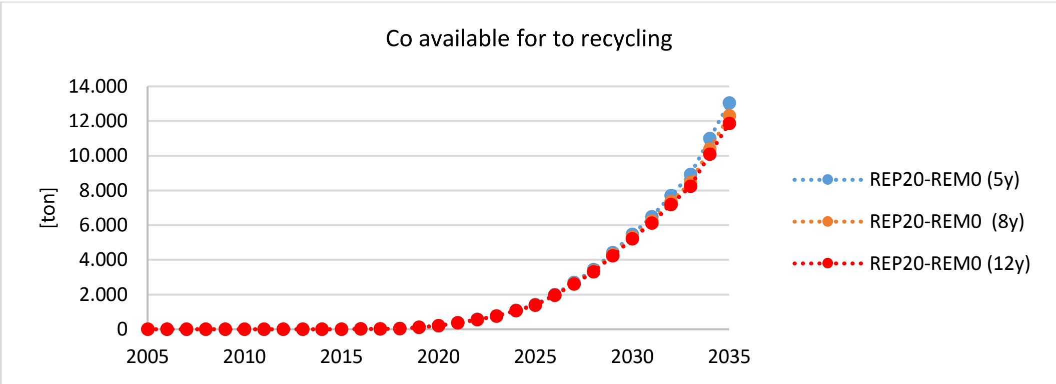
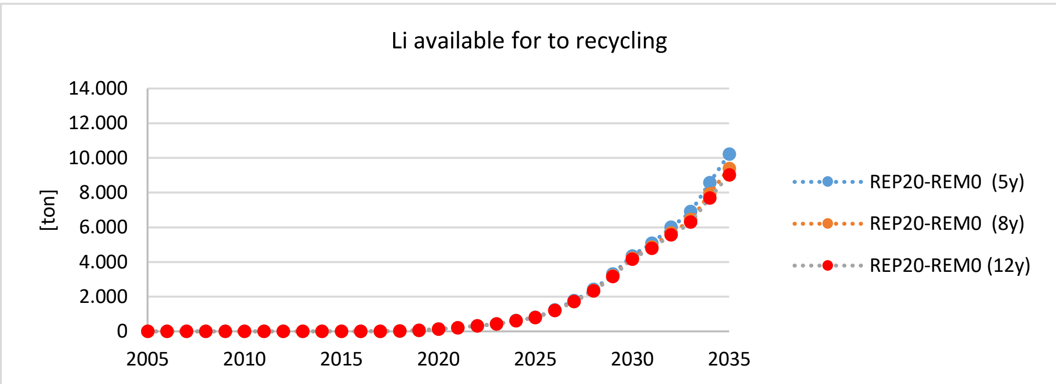



Figure 85: Variation of the Li available for recycling with the variation of the lifetime batteries during their second-use



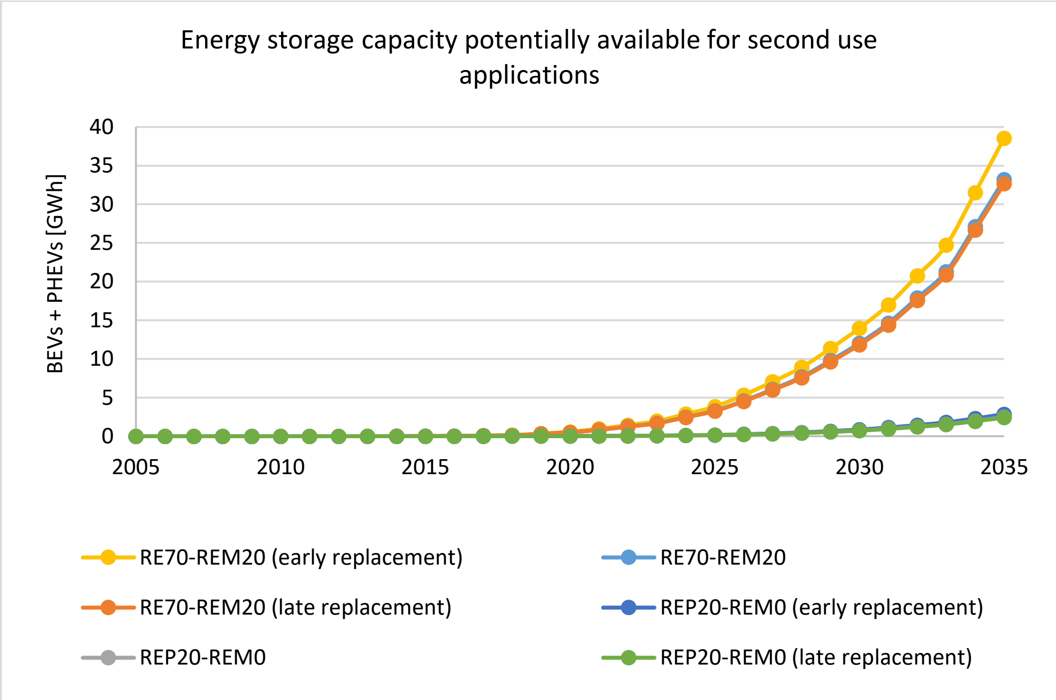

In order to assess the relevance of the residual capacity of LIBs when removed from xEVs, an "early replacement" and a "late replacement" are considered. For the "early replacement", it is assumed that

- the non-spent batteries (i.e. $\phi_{non\text{-}spent}$) are collected when their residual capacity is 90% of the nominal capacity,
- the spent batteries (i.e. $\phi_{maint}$ and $\phi_{spent}$) are collected at 70% of the nominal capacity.
- For the "late replacement",

- the non-spent batteries (i.e. $\phi_{spent}$) are collected when their residual capacity is 70% of the nominal capacity,
- the spent batteries (i.e. $\phi_{maint}$ and $\phi_{non\text{-}spent}$) are collected at 60% of the nominal capacity.

The variation of the energy storage capacity potentially available for second-use applications due to early replacement creates a significant increase only in the 'REP-70 scenario', which means that early replacement entails important energy savings compared to late replacement if second-use represents the main EoL option for LIBs. Compared to the "late replacement", the "early replacement" results in 2.14 GWh more to be potentially used in second-use applications in 2030 (increasing to 5.88 GWh in 2030) (Figure 86). Therefore, the "early replacement" could be an interesting option for utilities especially where high volumes of LIBs are adopted in second-use applications ('REP-70').

Figure 86: Variation of the energy storage capacity potentially available for second use applications with the variation of the lifetime of the second-used batteries




## 5.4 Lesson learnt and follow up

Extending lifetime of batteries is not yet developed in Europe and some of contacted stakeholders highlighted that the lack of a regulatory framework also entailing second-use of batteries is a barrier to the development of a business case related to second-use. However, the interest in the topic in proved by several stakeholders and some policy documents published in the last year. Sustainability of extending the lifetime of batteries should consider different aspects that all together can improve the management of the batteries' value-chain as a whole (technical, economic, social, environmental and legal). To estimate the impacts of extending the batteries' lifetime in terms of resources, a MFA model was developed to identify and estimate the flows and stocks of LIBs (and related aspects) in Europe. It contributes to increase the knowledge of the LIB value-chain in Europe.

As first step of the analysis, ad hoc questionnaires were developed to identify the main processes of the batteries value-chain in Europe. The obtained information from stakeholders were complemented with literature data in order to develop a MFA model describing the life-cycle steps and processes along the value-chain of LIBs after their removal from xEVs in Europe. In the model, all the possible EoL patterns (i.e. direct reuse, second-use and recycling) are captured (i.e. direct reuse, second-use and recycling).

Following the approach illustrated in chapter 5.2, parameters used in the model make it flexible and customisable according to the available input data and the interest of the users; different scenarios can be assessed and circular economy aspects and the effects of different EoL options can be identified. The main goal of the analysis is to identify the effects of the potential development of second-use of traction batteries in Europe in the next future. To have a more complete overview of second-use, the same MFA model is used also to estimate the stocks and flows of both the storage capacity related to batteries and the stocks and flows of relevant embedded materials (i.e. Co and Li).

The MFA model was applied to quantify the stocks and flows of both BEVs and PHEVs, the related storage capacity and the flows of Co and Li embedded in such batteries along the value-chain between 2005 and 2035. Three different scenarios were considered: second-use will not occur in Europe ('REP-0' scenario), second-use will progressively develop in Europe ('REP-20' scenario) and second-use will become the main EoL option in Europe ('REP-70' scenario).

Results pointed out that second-use allows a better exploitation of storage capacity of LIBs. On the other hand, recovery of cobalt and lithium to be recirculated in the European economy is delayed due to lifetime extension of LIBs. The relevance of this delay also depends on the development and deployment of recycling capacities at full-scale: the current high recycling rate of Co may contribute to a decrease in the demand for primary Co for LIBs; concerning Li, its potential recirculation cannot decrease the demand for lithium for LIBs as it is not currently recovered at industrial scale. Note that despite the recovery of materials could provide SRMs, recycling alone will not satisfy the request of raw materials for the batteries' manufacturing; therefore primary materials will be always needed (also according with Pavel and Blagoeva, 2017).

Based on this first analysis, different scenarios (including new 'REP-x scenarios') and the consequences of the development of various EoL patterns in Europe could be further assessed.

Overall, few detailed information on how the LIB market will evolve in Europe are publicly available. However, the relevance of the topic requires a more in-depth knowledge and the potential effects related to changes in the batteries value-chain (e.g. emerging of new EoL options as second-use). During the performed work, some relevant aspects emerged:

- The modelling of stocks and flows of a product in a specific system is relevant to better capture the specificity of the value-chain. The value-chain of LIBs in Europe could be different from the value-chain of LIBs in other geographical area, and also the involved stakeholder could be different. A strict collaboration with stakeholders in building the MFA model is an added value of such type of analysis
- The novelty of the topic and the confidentiality of some information have a huge effect of uncertainty of results. Emerging technologies and the fast development of the technology requires primary data and also expert judgement to understand the future changes of the products market. Sensitivity analysis and a constant update of the input data are recommended to improve the robustness of data. Moreover, the elaboration of data

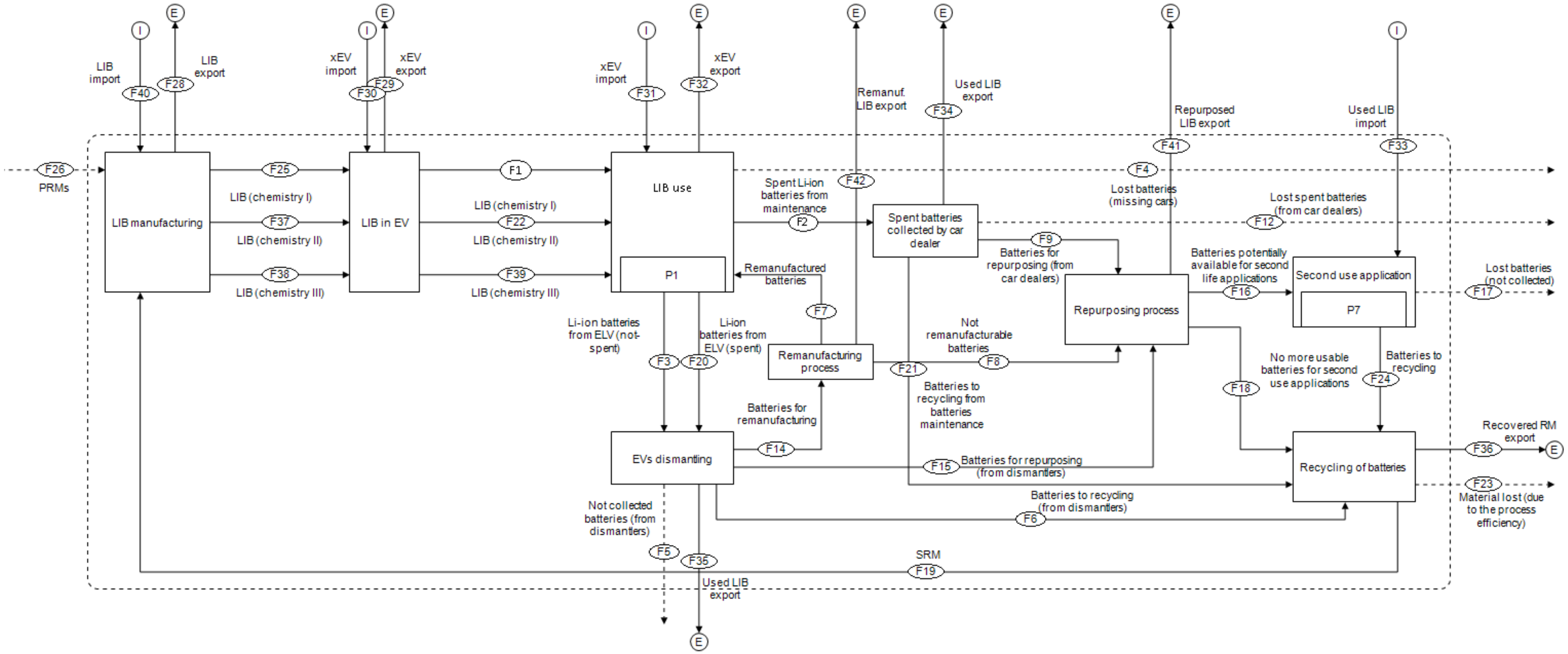
could provide more reliable data for the analysis. An example is the lifetime of LIBs in both first and second-life. Some authors suggested that the Weibull distribution could well model the lifetime of batteries (Rohr et al., 2017a).

- The possibility of creating different scenarios can enlarge the analysis and it is related to possible policy interventions (e.g. bans of some specific LIB chemistry or substance, minimum recycling content, re-use targets). This could also be analysed so that the model supports decision-making.

This MFA model can be used by a variety of actors to increase knowledge on second-use of batteries in Europe and to support the effective management of LIBs along their value-chain, e.g. collectors (to better organise collection schemes), for utilities (to estimate the overall capacity of LIBs that, after proper testing and repurposing can be potentially exploited in various applications), car manufacturers (to understand how to manage the possible development of a business case related to second-use).

In this perspective, the model should entail the whole batteries value-chain in order to enlarge the analysis and to have a complete overview of the stocks and flows. For instance, flows of import/export should be considered, as well as the extraction of materials and the manufacturing phase. The detail of the recycling step could ease the understanding of the flows of SRMs that could be recirculate in the economy and potentially decrease the demand (and the dependency) of RM for manufacturing batteries; it is to be noticed that, aligned to this statement, recycling of LIBs in Europe is expected to significantly increase after 2025 (Pavel and Blagoeva, 2017). An example of a MFA model with the extended system boundaries is depicted in Figure 87.

Figure 87: Potential evolution of the value-chain model of xEV batteries in Europe (see Figure 73)



The performed analysis mainly focused on second-use of batteries and consequent effect. Also remanufacturing of batteries contributes to extend the lifetime of batteries, and some examples of remanufactured batteries worldwide (Hanley, 2018; ReMaTec, 2018)[88] and in Europe already exist (e.g. SNAM, Peter Ursem) (Circusol Workshop (11/03/2019) and expert judgment BOX 2).

The model was applied to two materials embedded in batteries. However, it is to be noticed that other materials are important/critical for the EU, for instance natural graphite, manganese, nickel, copper. This model can be applied to other materials and track their stocks and flows (as both primary and secondary raw materials.
Finally, the MFA should be applied with other types of analysis (e.g. LCA) and other aspects, which have relevant effect on potential changes in the value-chain. Economic, legal and social aspects are important drivers for the development of a business case related to second-use of batteries.

[88] "In Japan 4R Energy Corporation, a Nissan and Sumitomo Corp JV, will reassemble a few hundred battery modules annually from the first-generation Nissan Leaf, where the used pack's overall energy capacity has fallen below 80%"

# Conclusions

Extending the lifetime of products through different strategies can entail benefits from different perspectives, environmental, economic and social. In fact, "reuse of products" (in a large sense) maximizes resource efficiency, minimizes wastage and boosts circularity in Europe (Circular Economy, Waste Framework Directive). Currently, in Europe, lifetime extension of products is a niche market but the interest in expanding this sector is arising at different levels (e.g. policy makers, consumers) and in various product's sectors.
The debate of the effective environmental benefits related to the extension of lifetime of products is still open, due to various aspects, e.g. the absence of a methodology able to quantify the impacts (benefits/drawbacks) of reuse, the different characteristics of products that affect the potential lifetime extension, the lack of knowledge related to various practices of reuse, the absence of a standardized definition of the terms related to "reuse" and "life".

Based on the available and scientifically recognised assessment tools, in the thesis it is illustrated a methodological framework to quantify the environmental impacts of extending the lifetime through various strategies in a circular economy context. The proposed framework consists of different methodological components to be combined according to the research needs, the characteristics of products and the available knowledge at the time of the analysis in order to assess multiple criteria. The framework was built thanks to the pro-active involvement of stakeholders and an extensive data collection; it contributes to the assessment of impacts of lifetime extension of both specific products (micro scale) and product groups (meso scale).
A Life Cycle Thinking approach is absolutely needed to have a holistic overview of the impacts of extending the products' lifetime. Modular Life Cycle Assessment (LCA) and the adoption of parameters were used to provide the necessary information of products/services and to make the LCAs flexible enough to facilitate the implementation of LCA models of different products considering the potential technological development (e.g. new components and/or new materials). LCA was combined with *Resource Efficiency Assessment of Products* (REAPro) method to assess the resource efficiency, too. Since the development and the strengthening of a market related to "reused products" depends also on economic aspects, on users awareness and on the quantity of reusable products, an economic assessment based on a life-cycle approach was combined to the LCA and REAPro methods to develop the *Environmental and Economic Assessment of Durability of Product* (Pro-EnDurAncE). Based on the built experience and on available studies in the scientific literature, to capture the effects of extending the lifetime of products, LCA is complemented by the analysis of the variation of stocks and flows of products and materials to provide a more complete understanding of products' status. This is particularly relevant in case of such analyses are required to support decision-making. Therefore, Material Flow Analysis (MFA) represents another methodological component of the proposed framework. Different stakeholders of the value chain of assessed products can benefit from the proposed methodological framework since it is flexible and customizable enough to identify aspects in which they are interested, and to better manage the whole life-cycle of products from an environmental perspective. This speeds-up the development of assessments of various products, permits updates according to the availability of data, offers the opportunity to take in account the change of technology and, according to the interest of users, could answer to questions related to future scenarios.

The proposed methodological framework is applied to two relevant sectors in Europe from an environmental perspective: house appliances and vehicles. For the house appliances sector, the analysis focused on vacuum cleaners (VCs) products group, whereas for vehicles, on passenger cars, with a special focus on traction batteries since they have a significant contribution to the environmental and economic impacts on electric vehicles (xEV). In both case studies, lack of data is addressed through the collection of primary data (e.g. from dismantling of products), through the interview with relevant stakeholders of the value chain of both products (e.g. development of *ad hoc* questionnaires) and through an extensive literature review (including scientific documents and grey literature[89]).
According to the characteristics of the case-study products, the Pro-EnDurAncE method is applied to the VC product group to quantify the environmental and economic impacts of extending the lifetime of VCs (Bobba et al., 2016a). For the automotive sector, the same approach was adopted and, thanks to the experts in the sector that were contacted during the developed work, it emerged the relevance of traction batteries as a component of xEV that can have an important contribution to the sustainability of xEV from different perspectives. Therefore, based on the characteristics of batteries, an adapted LCA was developed (Bobba et al., 2018a). To assess the effects of the lifetime extension of batteries, the performed analysis was coupled with a MFA aiming at assessing the variation of stocks and flows of batteries and related aspects. In particular, the developed dynamic MFA model is used to estimate the stocks and flows of traction batteries between 2005 and 2030 in Europe, the storage capacity and two different embedded materials (cobalt and lithium) (Bobba et al., 2019). In all cases, the adoption of parameters in all the developed analyses and the modularity of the developed models made the assessment models updatable and customizable according to users' needs. This also facilitate the performance of sensitivity analyses taking into account the uncertainty of input data.

Overall, results of the performed analyses pointed out that, extending the lifetime of both VCs and traction batteries, some environmental benefits occur even though different strategies are used (repair for VC and second-use for batteries).
Also, aspects related to the potential adoption of "reused" products emerged as important to be considered in the assessment, e.g. economic aspects and the user behaviour in the VC case-study (e.g. cost of new VCs, costs of auxiliaries, confidence in repaired products, availability of information for repairing VCs, etc.) and the geographical boundaries in case of the traction batteries (e.g. energy mix, degradation of the battery, adoption of batteries to store renewable energy, etc.).
Specificities of the assessed products heavily influenced the results of the assessments. This is particularly relevant in case of second-use of traction batteries, where an *ad hoc* model was used to estimate the energy flow of the system according to the characteristics of the batteries, the application and their relation.
The combination of different assessment tools emerged as a significant added value of the performed analysis. For instance, in case of VC, the contribution of dust bags to the life-cycle impacts emerged from the economic assessment and not from the environmental assessment. In case of traction batteries, the relevance of the lifetime of batteries emerged from both the adapted

[89] Grey literature stands for e.g. government and industrial reports, policy statements, consumers' forum, etc.

LCA and the MFA; however, the effects of second-use of batteries in terms of delay of recycling (and therefore of availability of secondary raw materials) emerged only from the MFA.

Focusing on household appliances, and VCs in particular, the obtained results of the durability assessment underlined that environmental benefits occur in particular for the impact categories that are mainly affected by manufacturing, e.g. abiotic depletion potential ($ADP_{res}$). For example, the lifetime extension of a VC by 100 hour (i.e. +20% for a lifetime equal to 600 hours) can potentially grant the saving of more than 20% $ADP_{res}$ compared to the replacement of the VC with a new one 15% more efficient. Some environmental effects also occur for the impact categories dominated by the use-phase (e.g. global warming potential - GWP): extending the lifetime of a VC by 100 hours saves around 1.5% of life-cycle GWP compared to the replacement of the VC with a new one 15% more efficient. It is to be noticed that, analysing the life-cycle costs of VCs, in case of extending the lifetime of VC for 100 hours, this brings a life-cycle saving of about 11-13 €, depending on the energy efficiency of the replacing VC. Higher benefits occur if the lifetime if further extended. Differently from the environmental assessment, it is observed that, independently of the energy efficiency of the replacing VC, the costs of maintenance and auxiliaries components significantly contributes to the life-cycle costs, being higher than 35% of the life-cycle costs.

Also in case of the assessment of the lifetime extension of traction batteries through their repurposing and second-use in different applications, results prove that some environmental benefits occur under certain conditions. In particular, higher benefits occur when repurposed batteries are coupled with renewable energy sources (i.e. increase the PV self-consumption rather than peak shaving in a grid-connected office building). To increase the PV self-consumption in a grid-connected house, the adoption of a repurposed battery in place of fresh battery allows a reduction of 94% of the life-cycle $ADP_{res}$ and 64% of the life-cycle GWP; in this case, the impacts of the manufacturing of the battery are fully allocated to the first life of the traction battery. If 25% of the manufacturing impacts are allocated to the second life, the reduction of respectively the life-cycle ADP-res and GWP decrease to 44% and 26%. No environmental benefits are observed in case the repurposed battery does not replace any batteries; on the contrary, in case of the adoption of a repurposed battery without replacing a fresh battery, but in a stand-alone configuration, allows a reduction of respectively 44% and 49% of the life-cycle $ADP_{res}$ and GWP. Not significant changes were observed from the performed sensitivity analyses (in case of replacement of some cells in the modules and higher energy for the battery repurposing).

The analysis of stocks and flows related to traction batteries in Europe permitted to observe that the second-use of traction batteries allows a better exploitation of the storage capacity of batteries. In particular, if second-use will gradually grow in Europe, 0.6 GWh of residual capacity can be further exploited in second-use applications in 2025, increasing to 2.7 GWh in 2030. Furthermore, lithium and cobalt embedded in batteries will last in the stock for around 5-10 years (depending on the second-use application), postponing their recovery and consequent availability as secondary raw materials. It is estimated that, in case of gradually development of second-use of batteries, in 2030 the cobalt embedded in batteries and available for recycling is about 5,000 tonnes, which is about 15% of the cobalt demand in 2030.

The developed work belongs to the field of resource efficiency, on which the interest of both business and policy stakeholders is currently increasing. The choice of the methodological components and their combination offers an overview of the complexity of the system and allows the assessment of multiple criteria, which are unlikely to be captured with a single assessment tool. Pros and cons of extending the products' lifetime compared to the recycling option requires an in-depth analysis able to balance the benefits of these two circular options. The analysis should consider both the specificities of the assessed product (as well as of its value-chain) and the system in which this product is used. As an example, extending the lifetime of products could maximize the resource efficiency of materials embedded in products but, on the other hand, their recovery and availability as secondary raw materials is delayed (which is particularly relevant in case of materials for which supply risks exist, e.g. critical raw materials - CRMs). Moreover, CRMs in EU are different from CRMs in China or US and the recycling of specific materials depends on the availability of technologies; how much is relevant the delay of SRMs compared to the demand of specific materials in a geographical area? The developed framework can be adopted to better understand the possible answers to this and other questions.

Further work should focus on the application of the proposed framework to enlarge the knowledge of the environmental, economic and social effects of extending the lifetime of products belonging to different sectors in a life-cycle perspective (e.g. infrastructures or packaging). This approach should be adopted especially in case the analyses' results are expected to support decision-making and effective management of EoL of products. To increase the knowledge on lifetime extension, the stakeholders' opinion should be used to test further the effectiveness and the robustness of the method and to implement it, also through the development of *ad hoc* scenarios to answer questions related to future scenarios.

Finally, the proposed framework can contribute to enlarge the assessment of extending the lifetime of products from a micro/meso scale to a macro scale, e.g. focusing on a specific geographical area, like the European economy, to capture the effects of options higher than recycling in the waste hierarchy.

# References


ABattReLife. Available at http://www.abattrelife.eu/ (accessed 11.1.16).

Abele, E., Anderl, R., Birkhofer, H., 2005. Environmentally-Friendly Product Development Methods and Tools. 6th ed., Springer. London.

ACEA, 2012. ACEA Position on Life Cycle Assessment. Available at http://www.acea.be/publications/article/position-paper-life-cycle-assessment.

ADEME, 2011. Etude de la seconde vie des batteries des vehicules electriques et hybrides rechargeables. ADEME (Agence de l'Environnement et de la Maîtrise de l'Energie). Angers (France).

AEA, 2009. Work on Preparatory Studies for Eco-Design Requirements of EuPs (II) Lot 17 Vacuum Cleaners. Oxfordshire.

Ahmadi, L., Fowler, M., Young, S.B., Fraser, R.A., Gaffney, B., Walker, S.B., 2014a. Energy efficiency of Li-ion battery packs re-used in stationary power applications. Sustain. Energy Technol. Assessments 8, 9–17. doi:10.1016/j.seta.2014.06.006

Ahmadi, L., Yip, A., Fowler, M., Young, S.B., Fraser, R.A., 2014b. Environmental feasibility of re-use of electric vehicle batteries. Sustain. Energy Technol. Assessments 6, 64–74. doi:10.1016/j.seta.2014.01.006

Allacker, K., Mathieux, F., Manfredi, S., Pelletier, N., De Camillis, C., Ardente, F., Pant, R., 2014. Allocation solutions for secondary material production and end of life recovery: Proposals for product policy initiatives. Resour. Conserv. Recycl. 88, 1–12. doi:10.1016/j.resconrec.2014.03.016

Alonso Raposo, M., Ciuffo, B., Ardente, F., Aurambout, J.-P., Baldini, G., Braun, R., Christidis, P., Christodoulou, A., Duboz, A., Felici, S., Ferragut, J., Georgakaki, A., Gkoumas, K., Grosso, M., Iglesias, M., Julea, A., Krause, J., Martens, B., Mathieux, F., Pekar, F., Raileanu, I.-C., Tamba, M., Tsakalidis, A., van Balen, M., Vandecasteele, I., 2019. The future of road transport - Implications of an automated, connected, low-carbon and shared mobility. EUR XXXXX EN, Publications Office of the European Union, Luxembourg.

Alpstore. Available at http://www.alpstore.info/ (accessed 11.1.16).

Alves Dias, P., Blagoeva, D., Pavel, C.C., Arvanitidis, N., 2018. Cobalt: demand-supply balances in the transition to electric mobility. doi:10.2760/97710

APRA Europe, 2012. Remanufacturing Terminology. Available at http://www.apra-europe.org/dateien/downloads/Reman_Term_Guideline_2012-03-06.pdf.

Ardente, F., Mathieux, F., 2014a. Environmental assessment of the durability of energy-using products: method and application. J. Clean. Prod. 74, 62–73. doi:10.1016/j.jclepro.2014.03.049

Ardente, F., Mathieux, F., 2014b. Identification and assessment of product's measures to improve resource efficiency: the case-study of an Energy using Product. J. Clean. Prod. 83, 126–141. doi:10.1016/j.jclepro.2014.07.058

Ardente, F., Talens Peiró, L., Mathieux, F., Polverini, D., 2018. Accounting for the environmental benefits of remanufactured products: Method and application. J. Clean. Prod. 198, 1545–1558. doi:10.1016/J.JCLEPRO.2018.07.012

Ardente, F., Tecchio, P., Bobba, S., Mathieux, F., 2017. Assessment of resource efficiency in a life cycle perspective: the case of reuse, in: Atti Del XI Convegno Della Rete Italiana LCA -

Resource Efficiency e Sustainable Development Goals: Il Ruolo Del Life Cycle Thinking. Siena, 22-23 Giugno 2017. ENEA. ISBN: 978-88-8286-352-4.

Argonne National Laboratory, 2014. GREET Life-Cycle model.

ARN, 2014. 2Bcycled investigating second life for Li-Ion batteries [WWW Document]. URL http://www.arn.nl/en/news/2bcycled-investigating-second-life-for-li-ion-batteries/, http://www.arn.nl/en/news/2bcycled-gezamenlijk-onderzoek-naar-een-tweede-leven-voor-accus-uit-elektrische-autos/ (accessed 11.1.16).

Aziz, T., Ketjoy, N., 2017. PV Penetration Limits in Low Voltage Networks and Voltage Variations. IEEE Acc 5, 16784–16792. doi:10.1109/ACCESS.2017.2747086

Baldé, C.P., Feng Wang, Ruediger Kuehr, 2016. Transboundary movements of used and waste electronic and electrical equipment: Estimates from the European Union using trade statistics.

Bank of America Merrill Lynch, 2016. Technology – Global Electric Vehicle Battery Handbook : Disruptive with strong growth.

Barba-Gutiérrez, Y., Adenso-Díaz, B., Lozano, S., 2008. Eco-Efficiency of Electric and Electronic Appliances: A Data Envelopment Analysis (DEA). Environ. Model. Assess. 14, 439–447. doi:10.1007/s10666-007-9134-2

Batteries2020. Batteries2020 [WWW Document]. URL http://www.batteries2020.eu/

Bauer, C., 2010. Ökobilanz von Lithium-Ionen Batterien. Paul Scherrer Institut, Labor für Energiesystem-Analysen (LEA), Villigen, Switz.

Bauer, C., Hofer, J., Althaus, H.-J., Del Duce, A., Simons, A., 2015. The environmental performance of current and future passenger vehicles: Life Cycle Assessment based on a novel scenario analysis framework. Appl. Energy. doi:10.1016/j.apenergy.2015.01.019

Berckmans, G., Messagie, M., Smekens, J., Omar, N., Vanhaverbeke, L., Mierlo, J. Van, 2017. Cost projection of state of the art lithium-ion batteries for electric vehicles up to 2030. Energies 10. doi:10.3390/en10091314

Berger, M., Warsen, J., Krinke, S., Bach, V., Finkbeiner, M., 2012. Water Footprint of European Cars: Potential Impacts of Water Consumption along Automobile Life Cycles 46, 4096–4099. doi:10.1021/es2040043

Berman, K., Carlson, R., Dziuba, J., Kackson, J., Hamilton, C., Sklar, P., 2018. The Lithium Ion Battery and the EV Market: The Science Behind What You Can't See. BMO Capital Markets report.

Bermudez, V., 2017. Electricity storage supporting PV competitiveness in a reliable and sustainable electric network. J. Renew. Sustain. Energy 9. doi:10.1063/1.4974851

BIO by Deloitte, 2013. Material-efficiency efficiency Ecodesign Report and Module to the Methodology for the Ecodesign of Energy-related Energy related Products (MEErP), Stakeholder Consultation.

Blagoeva, D.T., Aves Dias, P., Marmier, A., Pavel, C.C., 2016. Assessment of potential bottlenecks along the materials supply chain for the future deployment of low-carbon energy and transport technologies in the EU. Wind power, photovoltaic and electric vehicles technologies, time frame: 2015-2030. JRC-EC (Joint Research Centre - European Commission). Available at https://ec.europa.eu/jrc/en/publication/eur-scientific-and-technical-research-reports/assessment-potential-bottlenecks-along-materials-supply-chain-future-deployment-low-carbon. doi:10.2790/08169

Blengini, G.A., Di Carlo, T., 2010. Energy-saving policies and low-energy residential buildings: an LCA case study to support decision makers in Piedmont (Italy). Int. J. Life Cycle Assess. 15, 652–665. doi:10.1007/s11367-010-0190-5

Blomsma, F., Brennan, G., 2017. The Emergence of Circular Economy: A New Framing Around Prolonging Resource Productivity. J. Ind. Ecol. 21, 603–614. doi:10.1111/jiec.12603

Bobba, S., Ardente, F., Mathieux, F., 2016a. Environmental and economic assessment of durability of energy-using products: Method and application to a case-study vacuum cleaner. J. Clean. Prod. 137. doi:10.1016/j.jclepro.2016.07.093

Bobba, S., Ardente, F., Mathieux, F., 2015. Technical Support for Environmental Footprinting, Material Efficiency in Product Policy and the European Platform on LCA - Durability assessment of vacuum cleaners. JCR Science and Policy Report. JRC-EC (Joint Research Centre - European Commission). Available at publications.jrc.ec.europa.eu/repository/bitstream/JRC96942/lb-na-27512-en-n_.pdf. doi:10.2788/563222

Bobba, S., Farina, A., Blengini, G.A., Zanetti, M.C., Bonino, F., Lattore, M., 2016b. Life Cycle Assessment applied to vehicles' end-of-life: case study of FCA Panda and EU policy context, in: 5th International Conference on Industrial and Hazardous Waste Management. Crete (Gresse).

Bobba, S., Mathieux, F., Ardente, F., Blengini, G.A., Cusenza, M.A., Podias, A., Pfrang, A., 2018a. Life Cycle Assessment of repurposed electric vehicle batteries: an adapted method based on modelling energy flows. J. Energy Storage 19, 213–225. doi:10.1016/J.EST.2018.07.008

Bobba, S., Mathieux, F., Blengini, G.A., 2019. How will second-use of batteries affect stocks and flows in the EU? A model for traction Li-ion batteries. Resour. Conserv. Recycl. 145, 279–291. doi:10.1016/J.RESCONREC.2019.02.022

Bobba, S., Podias, A., Di Persio, F., Messagie, M., Tecchio, P., Cusenza, M.A., Eynard, U., Mathieux, F., Pfrang, A., 2018b. Sustainability Assessment of Second Life Application of Automotive Batteries (SASLAB): JRC Exploratory Research (2016-2017): Final technical report: August 2018. EUR 29321 EN, Publications Office of the European Union, Luxembourg, ISBN 978-92-79-92835-2; doi:10.2760/53624, JRC112543.

Bosch, 2016. Bosch Energy Storage Solutions [WWW Document]. URL https://boschenergystoragesolutions.com/en/blog/-/blog/4335273/batteries-of-electric-cars-for-a-robust-electricity-grid-bosch-cooperates-with-bmw-and-vattenfall (accessed 11.1.16).

Boustani, A., Sahni, S., Graves, S.C., Gutowski, T.G., 2010. Appliance Remanufacturing and Life Cycle Energy and Economic Savings, in: Proceedings of the 2010 IEEE International Symposium on Sustainable Systems & Technology, IEEE. Arlington, US.

Boyden, A., Soo, V.K., Doolan, M., 2014. The Environmental Impacts of Recycling Portable Lithium-Ion Batteries. 23rd CIRP Conf. Life Cycle Eng. doi:10.1016/j.procir.2016.03.100

Braun, M., Magnor, D., 2009. Photovoltaic self-consumption in Germany - Using lithium-ion storage to increase self- consumed photovoltaic energy, in: 24th European Photovoltaic Solar Energy Conference 2009. Proceedings of the International Conference held in Hamburg, 21-25 September 2009.

Bray, K., Conover, D., Kintner-meyer, M., Viswanathan, V., Ferreira, S., Rose, D., Schoenwald, D., 2012. Protocol for Uniformly Measuring and Expressing the Performance of Energy Storage Systems. PNNL-22010 Rev 2 / SAND2016-3078, Pacific Northwest National Laboratory, and Sandia National Laboratories, Richland, Washington, and Albuquerque. New Mexico (USA).

Brondi, C., Carpanzano, E., 2011. A modular framework for the LCA-based simulation of production systems. CIRP J. Manuf. Sci. Technol. 4, 305–312. doi:10.1016/J.CIRPJ.2011.06.006

Brook Lyndhurst, 2011. Public understanding of product lifetimes and durability ( 2 ): reuse of bulky items A research report completed for the Department for Environment , Food and Rural Affairs by Brook Lyndhurst. Reports for the UK Department for Environment Food and Rural Affairs (DEFRA). London, UK.

Brunner, P.H., Rechberger, H., 2004. Practical Handbook of Material Flow Analysis. Lewis Publishers, CRC Press LLC, 2000 N.W. Corporate Blvd., Boca Raton, Florida 33431, USA.

Bundesverband Sekundärrohstoffe und Entsorgung, 2015. Market Report Plastics [WWW Document]. URL http://plasticker.de/preise/marktbericht2_en.php?j=15&mt=9&quelle=bvse (accessed 12.1.15).

Busch, J., Steinberger, J.K., Dawson, D.A., Purnell, P., Roelich, K., 2014. Managing critical materials with a technology-specific stocks and flows model. Environ. Sci. Technol. 48, 1298–1305. doi:10.1021/es404877u

Canals Casals, L., Amante García, B., 2016. Assessing electric vehicles battery second life remanufacture and management. J. Green Eng. 6, 77–98. doi:10.13052/jge1904-4720.614

Canals Casals, L., García, B.A., Aguesse, F., Iturrondobeitia, A., 2015. Second life of electric vehicle batteries: relation between materials degradation and environmental impact. Int. J. Life Cycle Assess. doi:10.1007/s11367-015-0918-3

Castro, M.B.G., Remmerswaaland, J. a. M., Reuter, M. a., 2003. Life cycle impact assessment of the average passenger vehicle in the Netherlands. Int. J. Life Cycle Assess. 8, 297–304. doi:10.1007/BF02978922

CEC, 2015. Environmentally Sound Management of End-of-Life Batteries from Electric-Drive Vehicles in North America. Commission for Environmental Cooperation (CEC). 103 pp., Montreal (Canada).

Center for Sustainable Energy, 2016. Secondary Use Applications of Plug-in EV Lithium-Ion Batteries [WWW Document]. URL https://energycenter.org/program/secondary-use-applications-plug-ev-lithium-ion-batteries (accessed 11.1.16).

Chagnes, A., Pospiech, B., 2013. A brief review on hydrometallurgical technologies for recycling spent lithium-ion batteries. J. Chem. Technol. Biotechnol. 88, 1191–1199. doi:10.1002/jctb.4053

Chancerel, P., Marwede, M., Mathieux, F., Talens Peiró, L., 2016. Feasibility study for setting-up reference values to support the calculation of recyclability / recoverability rates of electr(on)ic products. JRC-EC (Joint Research Centre - European Commission). doi:10.2788/901715

Charter, M., Young, A., Kielkiewicz-Young, A., Belmane, I., 2001. Integrated product policy (IPP) and eco-product development (EPD), in: Proceedings - 2nd International Symposium on Environmentally Conscious Design and Inverse Manufacturing. The Centre for Sustainable Design, UK. doi:10.1109/ECODIM.2001.992445

Chlopek, Z., Lasocki, J., 2011. Comprehensive evaluation of the environmental hazard caused by the operation of automotive vehicles. Arch. Motoryz. Arch. Automot. Eng.

Chmura, A., 2016. Driving Towards Decarbonisation of Transport: Safety, Performance, Second life and Recycling of Automotive Batteries for e-Vehicles - 2nd life applications, in: Putting Science into Standards (PSIS) Workshop 22-23/09/2016. Petten (The Netherlands).

Ciocia, A., 2017. Optimal Power Sharing between Photovoltaic Generators, Wind Turbines, Storage and Grid to Feed Tertiary Sector Users. Doctoral Dissertation Doctoral Program in Electrical, Electronics and Communications Engineering (29th Cycle). Politecnico di Torino (IT).

Cobb, J., 2014. How Long Will An Electric Car's Battery Last? [WWW Document]. URL https://www.hybridcars.com/how-long-will-an-evs-battery-last/

COMEX - CME Group, 2015. Aluminum Monthly Price [WWW Document]. URL http://www.indexmundi.com/commodities/?commodity=aluminum&months=12 (accessed 12.1.15).

Cooper, D.R., Gutowski, T.G., 2015. The Environmental Impacts of Reuse: A Review. J. Ind. Ecol. 21, 38–56. doi:10.1111/jiec.12388

Cooper, T., 2010. Longer Lasting Products: Alternatives To The Throwaway Society. Gower Publishing, Ltd. doi:978-0-566-08808-7

Cooper, T., 2005. Slower Consumption. Reflections on Product Life Spans and the "Throwaway Society." J. Ind. Ecol. 9.

Cooper, T., 1996. Beyond Recycling. The longer life option. The new economics foundations, London, UK.

Cready, E., Lippert, J., Pihl, J., Weinstock, I., Symons, P., 2003. Technical and Economic Feasibility of Applying Used EV Batteries in Stationary Applications, SAND Report, SAND2002-4084. SAND2010-0815. Sandia National Laboratories, Albuquerque, New Mexico 87185 and Livermore, California 94550 (USA).

Cusenza, M.A., Bobba, S., Ardente, F., Cellura, M., Di Persio, F., 2019. Energy and environmental assessment of a traction lithium-ion battery pack for plug-in hybrid electric vehicles. J. Clean. Prod. 2015, 634–649. doi:10.1016/j.jclepro.2019.01.056

Daimler, 2015. Daimler Sustainability report 2014. Available at http://sustainability.daimler.com/.

Dave, S., 2010. Life Cycle Assessment of Transportation Options for Commuters.

De Almeida, A., Falkner, H., Fong, J., 2013. EuP Lot 30: Electric Motors and Drives - Task 4: Technical analysis existing products.

De Almeida, A.T., Ferreira, F.J.T.E., Fong, J., Fonseca, P., 2008. EUP Lot 11 Motors Final Report. Coimbra.

De Meester, S., Nachtergaele, P., Debaveye, S., Vos, P., Dewulf, J., 2019. Using material flow analysis and life cycle assessment in decision support: A case study on WEEE valorization in Belgium. Resour. Conserv. Recycl. 142, 1–9. doi:10.1016/J.RESCONREC.2018.10.015

Debnath, U.K., Ahmad, I., Habibi, D., 2014. Quantifying economic benefits of second life batteries of gridable vehicles in the smart grid. Int. J. Electr. Power Energy Syst. 63, 577–587. doi:10.1016/J.IJEPES.2014.05.077

Deflorian, F., 2008. Materiali, risorse e ambiente: le tecnologie di riciclo dei materiali, in: Nuovi Materiali per Le Tecnologie Del Futuro. Facoltà di Ingegneria dell'Università degli Studi di Trento.

Delogu, M., Berzi, L., 2011. Metodologie ed applicazioni di LCA al settore automobilistico, in: Agenzia nazionale per le nuove tecnologie, l'energia e lo sviluppo economico (ENEA) (Ed.), Cappellaro F. and Scalbi S. in "La Rete Italiana LCA: Prospettive e Sviluppi Del Life Cycle Assessment in Italia."

Deloitte, 2015. Energy storage: Tracking the technologies that will transform the power sector. Available at http://www.ourenergypolicy.org/energy-storage-tracking-the-technologies-that-will-transform-the-power-sector/.

Delucchi, M., 2003. A Lifecycle Emissions Model (LEM): Lifecycle Emissions from Transportation Fuels, Motor Vehicles, Transportation Modes, Electricity Use, Heating and Cooking Fuels, and Materials.

DeRousseau, M., Gully, B., Taylor, C., Apelian, D., Wang, Y., 2017. Repurposing Used Electric Car Batteries: A Review of Options. Jom 69, 1575–1582. doi:10.1007/s11837-017-2368-9

Dewulf, W., Duflou, J.R., 2004. The environmentally optimised lifetime: A crucial concept in life cycle engineering. Proc. Glob. Conf. Sustain. Prod. Dev. Life Cycle Eng. 2004, 59–62.

Directive 1999/94/EC, 1999. This document is meant purely as a documentation tool and the institutions do not assume any liability for its contents. Off. J. Eur. Communities L 269, 1–15. doi:2004R0726 - v.7 of 05.06.2013

Directive 2009/33/CE, 2009. 15.5.2009. Off. J. Eur. Communities.

Donat Marques, C., 2018. Nickel? Electrification may boost demand [WWW Document]. URL https://www.axier.se/analys?commentid=342995 (accessed 1.3.19).

Downes, J., B. Thomas, Dunkerley, C., Walker, H., 2011. Longer Product Lifetimes: Summary Report. UK Dartment for Environment, Food and Rural Affairs (DEFRA).

Duce, A. Del, Egede, P., Öhlschläger, G., Dettmer, T., Althaus, H.-J., Bütler, T., Szczechowicz, E., 2013. Guidelines for the LCA of electric vehicles (eLCAr). E-Mobility Life Cycle Assessment Recommendations.

Dunn, J.B., Gaines, L., Barnes, M., Sullivan, J., Wang, M.Q., 2012. Material and energy flows in the materials production, assembly, and end-of-life stages of the automative lithium-ion battery life cycle. ARGONNE National laboratory (ANL). Available at http://greet.es.anl.gov/publication-li.

EATON. EATON - Powering Business Worldwide [WWW Document]. URL http://www.eaton.eu/Europe/OurCompany/News/PRCompany/PCT_2967726 (accessed 11.1.16).

EC, 2018a. Commission Staff working document. Measuring progress towards circular economy in the European Union – Key indicators for a monitoring framework. SWD(2018) 17 final. doi:10.2873/85546

EC, 2018b. Strategic Action Plan on Batteries. Annex 2 to the COM(2018) 293 final. COM(2018) 293 final.

EC, 2017a. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions on the 2017 list of Critical Raw Materials for the EU. COM(2017) 490 final.

EC, 2017b. Commission Staff working document Accompanying the document Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions EUROPE ON THE MOVE An agenda for a socially. SWD(2017) 177 final.

EC, 2017c. BATTERIES - a major opportunity for a sustainable society. European Commission (EC) Directorate-General for Research and Innovation. doi:10.2777/412120

EC, 2015. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions. Closing the loop - An EU action plan for the Circular Economy. COM(2015) 614 final.

EC, 2014a. Commission communication in the framework of the implementation of Commission Delegated Regulation (EU) No 665/2013 supplementing Directive 2010/30/EU of the European Parliament and of the Council with regard to energy labelling of vacuum cleaners and of . 2014/C 272/06.

EC, 2014b. Communication from the Commission to the Council and the European Parliament, the Council, the European economic and social Committee and the Committee of the Regions. Energy prices and costs in Europe (No. COM(2014) 21 /2). SWD(2014) 20 final.

EC, 2014c. Frequently Asked Questions on Directive 2006/66/EU on Batteries and Accumulators and Waste Batteries and Accumulators.

EC, 2013. Commission staff working document impact assessment Accompanying the documents: Commission Regulation implementing Directive 2009/125/EC and Directive 2010/30/EU. SWD(2013)240.

EC, 2011a. Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: Roadmap to a Resource Efficient Europe. COM (2011) 571.

EC, 2011b. Roadmap to a Single European Transport Area: Towards a Competitive and Resource Efficient Transport System. Publications Office of the European Union, COM(2011) 144 final.

EC, 2009. Commission Staff Working Document acoompanying he Report from the Commission to the Council, the European Parliament, the European and Social Committee and the Committee of the regions On the State of Implementation of Integrated Product Policy. SEC(2009)1707 final.

EC, 2008. Communication from the Commission to the Council and the European Parliament. Establishment of the working plan for 2009-2011 under the Ecodesign Directive (No. COM(2008) 660). COM(2008) 660 final.

EC, 2003. Communication COM(2003:302) - Integrated Product Policy - Building on Environmental Life-Cycle Thinking. COM(2003) 302 final.

EC, 2001. Green Paper on Integrated Product Policy. COM(2001) 68 final.

EC - JRC, 2012. Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. JRC-EC (Joint Research Centre - European Commission). doi:10.2788/60825

EC - JRC, 2011. ILCD Handbook: Recommendations for Life Cycle Impact Assessment in the European context. doi:10.278/33030

EC - JRC, 2010a. International Reference Life Cycle Data System (ILCD) Handbook - Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. Publications Office of the European Union, European Commission - Joint Research Centre - Institute for Environment and Sustainability. doi:10.2788/38479

EC - JRC, 2010b. ILCD Handbook: Framework and requirements for LCIA models and indicators. doi:10.2788/38719

Ecobilan S.A., 1996. Overview of Life Cycle Assessment Studies in the Automotive Sector.

EEA, 2018a. Waste prevention in Europe: policies, status and trends in 2017. European Environment Agency (EEA). Copenhagen (Denmark).

EEA, 2018b. Electric vehicles from life cycle and circular economy perspectives. doi:10.2800/77428

EEA, 2016. Electric Vehicles in Europe. European Environment Agency (EEA). Copenhagen(Denmark). doi:10.2800/100230

Elkind, E., 2014. Reuse and repower. How to save money and clean the grid with second-life electric vehicle batteries. Climate Change and Business Research Initiative at the UCLA School of Law's Emmett Institute on Climate Change and the Environment and UC Berkeley School of Law's Center for Law, Energy & the Environment (CLEE). Berkeley (CA).

Ellingsen, L.A.W., Majeau-Bettez, G., Singh, B., Srivastava, A.K., Valoen, L.O., Stromman, A.H., 2014. Life Cycle Assessment of a Lithium-Ion Battery Vehicle Pack. J. Ind. Ecol. 18, 113–124. doi:10.1111/jiec.12072

ELSA, 2017. Energy Local Storage Advanced system (ELSA) [WWW Document]. URL http://www.elsa-h2020.eu/ (accessed 11.1.16).

Eminton, S., Quinault, C., Date, W., 2015. Glass [WWW Document]. URL http://www.letsrecycle.com/prices/glass/glass-prices-2015/ (accessed 12.1.15).

EU, 2014. REGULATION (EU) No 333/2014 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL - the modalities for reaching the 2020 target to reduce CO2 emissions from new passenger cars.

EU, 2013a. Commission Regulation (EU) No 666/2013 of 8 July 2013 implementing Directive 2009/125/EC of the European Parliament and of the Council with regard to ecodesign requirements for vacuum cleaners.

EU, 2013b. Commission Delegated Regulation (EU) No 665/2013 of 3 May 2013 supplementing Directive 2010/30/EU of the European Parliament and of the Council with regard to energy labelling of vacuum cleaners.

EU, 2012. Directive 2012/19/EU of the European Parliament and of the Council of 4 July 2012 on waste electrical and electronic equipment (WEEE).

EU, 2010. Directive 2010/30/EU of the European Parliament and the Council of 19 May 2010 on the indication by labelling and standard product information of the consumption of energy and other resources by energy-related products.

EU, 2009. Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC. OJ L140/16.

EU, 2008. Directive 2008/98/EC of the European parliament and of the Council of 19 November 2008 on waste and repealing certain Directives. OJ L 312.

EU, 2006. Directive 2006/66/EC of the European Parliament and of the Council of 6 September 2006 on batteries and accumulators and waste batteries and accumulators and repealing Directive 91/157/EEC. OJ L 266.

EU, 2000. Directive 2000/53/EC of the European Parliament and of the Council of 18 September 2000 on end-of life vehicles. OJ L 269.

EUROBAT, 2016. Battery energy storage in the EU. Barriers, opportunities, services and benefits. Eurobat.Org.

EUROBAT, 2015. E-mobility battery R&D roadmap 2030 - Battery technology for vehicle applications. Association of European Automotive and Industrial Battery Manufacturers (EUROBAT). Available at https://eurobat.org/sites/default/files/e-mobility_roadmap_presentation_olivier_amiel_final.pdf.

EUROBAT, 2014. A review of battery technologies for automotive applications. Association of European Automotive and Industrial Battery Manufacturers (EUROBAT). Available at http://www.eurobat.org/substitution-battery-technologies-wouldimpact-significantly-passenger-cars-says-new-study-19-may-20.

EUROFER - the European Steel Association, 2015. Scrap price index [WWW Document]. URL http://www.eurofer.org/Facts&Figures/Scrap price index.fhtml (accessed 12.1.15).

European Alliance, 2011. Sustainability aspects of SMC / BMC based products.

European Commission, 2019. Reflection Paper - Towards a Sustainable Europe By 2030. Available at https://ec.europa.eu/commission/sites/beta-political/files/rp_sustainable_europe_30-01_en_web.pdf

European Commission, 2018. European Battery Alliance [WWW Document]. URL https://ec.europa.eu/growth/industry/policy/european-battery-alliance_en

European Commission, 2017. Batteries - A major opportunity for a sustainable society - Research & Innovation Projects for Policy. European Commission, Luxembourg. doi:10.2777/864893

European Parliament, 2009. Directive 2009/125/EC of the European Parliament and of the Council of 21 October 2009 establishing a framework for the setting of ecodesign requirements for energy-related products (recast). Off. J. Eur. Union.

European Union, 2017. Workshop Report: Promoting Remanufacturing, Refurbishment, Repair and Direct Reuse.

EUROSTAT, 2018. Passenger cars in the EU [WWW Document]. URL https://ec.europa.eu/eurostat/statistics-explained/index.php/Passenger_cars_in_the_EU#Overview

EUROSTAT, 2015. Electricity and natural gas price statistics [WWW Document]. URL http://ec.europa.eu/eurostat/statistics-explained/index.php/Electricity_and_natural_gas_price_statistics#Electricity_prices_for_household_consumers (accessed 12.1.15).

Eyer, J., Corey, G., 2010. Energy Storage for the Electricity Grid: Benefits and Market Potential Assessment Guide. SAND2010-0815. Sandia National Laboratories, Albuquerque, New Mexico 87185 and Livermore, California 94550 (USA).

Eynard, U., Bobba, S., Cusenza, M.A., Blengini, G.A., 2018. Lithium-ion batteries for electric vehicles: combining Environmental and Social Life Cycle Assessments, in: Life Cycle Thinking in Decision-Making for Sustainability: From Public Policies to Private Businesses. Messina, 11th and 12th June 2018.

Faria, R., Marques, P., Garcia, R., Moura, P., Freire, F., Delgado, J., De Almeida, A.T., 2014. Primary and secondary use of electric mobility batteries from a life cycle perspective. J. Power Sources 262, 169–177. doi:10.1016/j.jpowsour.2014.03.092

Fastmarkets, 2015. Base Metals Charts [WWW Document]. URL http://www.fastmarkets.com/base-metals/base-metals-charts (accessed 12.1.15).

FEFCO, 2011. European Database for Corrugated Board Life Cycle Studies.

Finkbeiner, M., Hoffmann, R., Ruhland, K., Liebhart, D., Stark, B., 2006. Application of Life Cycle Assessment for the Environmental Certificate of the Mercedes-Benz S-Class. Int. J. Life Cycle Assess. 11, 240–246. doi:http://dx.doi.orgl1 0.1 085Jica2008.05.248

Fischer-Kowalski, M., Krausmann, F., Giljum, S., Lutter, S., Mayer, A., Bringezu, S., Moriguchi, Y., Schütz, H., Schandl, H., Weisz, H., 2011. Methodology and indicators of economy-wide material flow accounting: State of the art and reliability across sources. J. Ind. Ecol. 15, 855–876. doi:10.1111/j.1530-9290.2011.00366.x

Forsee Power, 2015. Battery Second Life Project [WWW Document]. URL https://www.edf.fr/en/the-edf-group/dedicated-sections/press/all-press-releases/battery-second-life-project (accessed 12.1.16).

Friedrich, B., Peters, L., 2017. Status and Trends of industrialized Li-Ion battery recycling processes with qualitative comparison of economic and environmental impacts, in: 22nd ICBR - International Congress on Battery Recycling.

Frischknecht, R., Jungbluth, N., Althaus, H., Bauer, C., Doka, G., Dones, R., Hischier, R., Hellweg, S., Köllner, T., Loerincik, Y., Margni, M., Nemecek, T., 2007. Implementation of Life Cycle Impact Assessment Methods. No. 3, v2.0. Swiss Centre for Life Cycle Inventories, Dübendorf.

Gabrisch, C., Cerdas, F., Herrmann, C., 2019. Product System Modularization in LCA Towards a Graph Theory Based Optimization for Product Design Alternatives, in: Progress in Life Cycle Assessment. Springer, pp. 37–44.

Gaines, L., Cuenca, R., 2000. Costs of Lithium-Ion Batteries for Vehicles. Energy 48, 73. doi:10.1002/art.10781

Gandy, S., Dolley, P., Parry, G., 2012. Developing an evidence base on flushing toilets and urinals - Final Report, Task 4.

Gasparin, N., 2015. Automotive Battery Market Outlook - Update 2015. A report of the EUROBAT Automotive Battery Committee.

Gharfalkar, M., Ali, Z., Hillier, G., 2016. Clarifying the disagreements on various reuse options: Repair, recondition, refurbish and remanufacture. Waste Manag. Res. 34, 995–1005. doi:10.1177/0734242X16628981

Goedkoop, M., Heijungs, R., Huijbregts, M.A.J., De Schryver, A., Struijs, J., V.Z., R., 2009. ReCiPe 2008 A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Report I: Characterisation factors, first edition.

Goedkoop, M. and Spriensma, R. (2000). Ministry of VROM, T., Hague, T.N. (updated version: www. pre. nl/eco-indicator9. ), 2000. No TThe Eco-indicator 99: A damage oriented method for Life Cycle Impact Assessment Methodology.itle.

Goedkoop, M., Huijbregts, M., 2012. ReCiPe 2008.

Gordon-Bloomfield, N., 2015. Nissan, 4R Energy, Green Charge Networks, Announce Second-Life Grid-Tied Storage Program for Electric Car Battery Packs [WWW Document]. URL https://transportevolved.com/2015/06/16/nissan-4r-energy-green-charge-networks-announce-second-life-grid-tied-storage-program-for-electric-car-battery-packs (accessed 11.1.16).

Görtz, S., 2015. Battery energy storage for intermittent renewable electricity production: A review and demonstration of energy storage applications permitting higher penetration of renewables. UMEA Universitet. Available at http://www.diva-portal.org/smash/record.jsf?pid=diva2%3A818683&dswid=-4677.

Groen, B. in 't, 2016. Driving Towards Decarbonisation of Transport: Safety, Performance, Second life and Recycling of Automotive Batteries for e-Vehicles - Second life applications, in: Putting Science into Standards (PSIS) Workshop 22-23/09/2016. Petten (The Netherlands).

Gruber, P.W., Medina, P.A., Keoleian, G.A., Kesler, S.E., Everson, M.P., Wallington, T.J., 2011. Global lithium availability: A constraint for electric vehicles? J. Ind. Ecol. 15, 760–775. doi:10.1111/j.1530-9290.2011.00359.x

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., Van Oers, L., Anneke, W.S., Suh, S., De Haes, U.H.A., De Bruijn, H., Van Duin, R., Huijbregts, M.A.J., 2001. Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards, 2001. Centre of Environmental Science. Leiden University, School of Systems Engineering, Policy Analysis and Management – Delft University of Technology, Fuels and Raw Materials Bureau, Interfaculty Department of Environmental Science, University of Amsterdam, Department of Environmental Studi.

Hanley, S., 2018. Nissan Begins Offering Remanufactured Batteries For LEAF [WWW Document]. URL https://cleantechnica.com/2018/05/15/nissan-begins-offering-remanufactured-batteries-for-leaf/ (accessed 2.2.19).

Hartwell, I., Marco, J., 2016. Management of intellectual property uncertainty in a remanufacturing strategy for automotive energy storage systems. J. Remanufacturing 6, 3. doi:10.1186/s13243-016-0025-z

Haupt, M., Kägi, T., Hellweg, S., 2018. Modular life cycle assessment of municipal solid waste management. Waste Manag. 79, 815–827. doi:10.1016/J.WASMAN.2018.03.035

Hawkins, T.R., Gausen, O.M., Strømman, A.H., 2012. Environmental impacts of hybrid and electric vehicles-a review. Int. J. Life Cycle Assess. 17, 997–1014. doi:10.1007/s11367-012-0440-9

Hawkins, T.R., Singh, B., Majeau-Bettez, G., Strømman, A.H., 2013. Comparative Environmental Life Cycle Assessment of Conventional and Electric Vehicles. J. Ind. Ecol. 17, 53–64. doi:10.1111/j.1530-9290.2012.00532.x

Heymans, C., Walker, S.B., Young, S.B., Fowler, M., 2014. Economic analysis of second use electric vehicle batteries for residential energy storage and load-levelling. Energy Policy 71, 22–30. doi:10.1016/j.enpol.2014.04.016

Hinterstocker, M., Schott, P., Roon, S. Von, Energiewirtschaft, F., Blütenanger, A., 2017. Increasing residential self-consumption of PV energy by DSM 6–11.

Horie, Y.A., 2004. Life Cycle Optimization of Household Refrigerator-Freezer Replacement. Master's Thesis. University of Michigan.

Huo, H., Cai, H., Zhang, Q., Liu, F., He, K., 2015. Life-cycle assessment of greenhouse gas and air emissions of electric vehicles: A comparison between China and the U.S. Atmos. Environ. 108, 107–116. doi:10.1016/j.atmosenv.2015.02.073

Hur, T., Lee, J., Ryu, J., Kwon, E., 2005. Simplified LCA and matrix methods in identifying the environmental aspects of a product system. J. Environ. Manage. 75, 229–37. doi:10.1016/j.jenvman.2004.11.014

Huss, A., Maas, H., Hass, H., 2013. TANK-TO-WHEELS Report Version 4 . 0 JEC WELL-TO-WHEELS ANALYSIS. doi:10.2788/40409

Huysman, S., Debaveye, S., Schaubroeck, T., De Meester, S., Ardente, F., Mathieux, F., Dewulf, J., 2015. Recyclability benefit rate indicator upgraded for open-loop recycling and applied for estimation of resource benefit of plastic waste management in Flanders. Resour. Conserv. Recycl. 101, 53–60.

ICCT, 2018. Effects of battery manufacturing on electric vehicle life-cycle greenhouse gas emissions | International Council on Clean Transportation.

IEA, 2018. Global EV Outlook 2018: Towards cross-modal electrification. OECD, International Energy Agency (IEA). doi:10.1787/9789264302365-en

IEC, 2012. IEC/TR 62635 - Guidelines for end-of-life information provided by manufacturers and recyclers and for recyclability of electrical and electronic equipment. Edition 1.0 2010-09. Geneva, Switzerland.

Igos, E., Benetto, E., Meyer, R., Baustert, P., Othoniel, B., 2018. How to treat uncertainties in life cycle assessment studies? Int. J. Life Cycle Assess. 1–14. doi:10.1007/s11367-018-1477-1

IHS Consulting, 2014. The availability of automotive lead-based batteries for recycling in the EU. Available at www.eurobat.org/sites/default/files/ihs_eurobat_report_lead_lores_final.pdf.

InfoMine, 2015a. Copper price [WWW Document]. URL http://www.infomine.com/ChartsAndData/ChartBuilder.aspx?z=f&gf=110563.USD.lb&dr=5y&cd=1 (accessed 12.1.15).

InfoMine, 2015b. Zinc Price [WWW Document]. URL http://www.infomine.com/ChartsAndData/ChartBuilder.aspx?z=f&gf=110565.USD.lb&dr=1y&cd=1 (accessed 12.1.15).

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories 2, 12. doi:http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/2_Volume2/V2_3_Ch3_Mobile_Combustion.pdf

Iraldo, F., Facheris, C., 2015. DRAFT Report. The Durability of Products - Standard assessment for the circular economy under the Eco-Innovation Action Plan. Task 4 – Cost Benefit Analysis of Using More Durable Products. Didcot, UK.

ISO, 2006a. ISO 14040:2006. Environmental management - Life cycle assessment - Principles and framework.

ISO, 2006b. ISO 14044:2006. Environmental management - Life cycle assessment - Requirements and guidelines.

ISO 14044:2006, 2006. ISO 14040:2006. Environmental management - Life cycle assessment - Requirements and guidelines.

Itsubo, N. et al. (2008a), 2008. LIME documentation. Eutrophication. Chapter 2.8 of draft English translation of LIME documentation (figures and tables in Japanese).

James Paul, Liss, J., Holmes, J., Sohnen, J., 2015. The market impact of standardized design in commercial PEV battery pack purchase and disposal. Prepared for the California Energy Commission (CEC-500-2016-028).

Jamesh, M.I., Prakash, A.S., 2018. Advancement of technology towards developing Na-ion batteries. J. Power Sources 378, 268–300. doi:10.1016/J.JPOWSOUR.2017.12.053

Johnson, J.X., Mcmillan, C.A., Keoleian, G.A., 2013. Evaluation of life cycle assessment recycling allocation methods: The case study of aluminum Johnson et al. Evaluation of LCA allocation methods. J. Ind. Ecol. 17, 700–711. doi:10.1111/jiec.12050

Johnston, K., 2013. The Life Expectancy Of Appliances [WWW Document]. URL http://activerain.trulia.com/blogsview/4181872/the-life-expectancy-of-appliances (accessed 2.1.16).

Jolliet, O., Müller-Wenk, R., Bare, J., Brent, A., Goedkoop, M., Heijungs, R., Itsubo, N., Peña, C., Pennington, D., Potting, J., Rebitzer, G., Stewart, M., Haes, H.U., Weidema, B., 2004. The LCIA midpoint-damage framework of the UNEP/SETAC life cycle initiative. Int. J. Life Cycle Assess. 9, 394–404. doi:10.1007/BF02979083

Joulié, M., Laucournet, R., Billy, E., 2014. Hydrometallurgical process for the recovery of high value metals from spent lithium nickel cobalt aluminum oxide based lithium-ion batteries. J. Power Sources 247, 551–555. doi:10.1016/J.JPOWSOUR.2013.08.128

JRC, 2013. Critical Metals in the Path towards the Decarbonisation of the EU Energy Sector. JRC Scientific & Policy Report, European Commission. doi:10.2790/46338

Kampman, B., Essen, H. Van, Braat, W., Grünig, M., Kantamaneni, R., Gabel, E., 2011. Impacts of Electric Vehicles - scenarios and policy implications 165.

Kane, M., 2016. BMW and Bosch's Second-Life 2.8 MWh Energy Storage Solution in Hamburg [WWW Document]. URL http://insideevs.com/bmw-and-boschs-second-life-2-8-mwh-energy-storage-solution-in-hamburg/ (accessed 11.1.16).

Kemna, R., 2011a. MEErP 2011 - Methodology Report. Part 2: Environmental policies & data. Van Holsteijn en Kemna B.V. (VHK) - Brussels/Delft.

Kemna, R., 2011b. MEErP 2011 - Methodology Report. Part 1: Methods. Van Holsteijn en Kemna B.V. (VHK). Delft, Netherlands.

Kempener, R., Borden, E., 2015. Battery Storage for Renewables: Market Status and Technology Outlook. International Renewable Energy Agency (IRENA).

Kenma, R., van Elburg, M., Li, W., van Holsteijn, R., 2005. MEEUP Methodology Report. Van Holsteijn en Kemna B.V. (VHK). Delft, Netherlands.

Kessels, K., Mantels, B., Hussy, C., Bons, M., Comaty, F., Goes, M., Wiersma, F., Kshemendranat, A., Christensen, C., Hochloff, P., Schledde, D., 2017. Support to R & D Strategy for battery based energy storage. Costs and benefits for deployment scenarios of battery systems. BATSTORM project - D7.

Kim, H.C., Wallington, T.J., Arsenault, R., Bae, C., Ahn, S., Lee, J., 2016. Cradle-to-Gate Emissions from a Commercial Electric Vehicle Li-Ion Battery: A Comparative Analysis. Environ. Sci. Technol. 50, 7715–7722. doi:10.1021/acs.est.6b00830

Klör, B., Bräuer, S., Beverungen, D., 2014. A Business Process Model for the Reverse Logistics of Used Electric Vehicle Batteries. Proc. 44. Jahrestagung der Gesellschaft für Inform. 5, 1631–1644.

Kobayashi, Y., Kobayashi, H., Hongu, A., Sanehira, K., 2005. A Practical Method for Quantifying Eco-efficiency Using Eco-design Support Tools. J. Ind. Ecol. 9, 131–144. doi:10.1162/108819805775247990

Koch-Ciobotaru, C., Saez-de-Ibarra, A., Martinez-Laserna, E., Stroe, D.-I., Swierczynski, M., Rodriguez, P., 2015. Second life battery energy storage system for enhancing renewable energy grid integration. Energy Convers. Congr. Expo. (ECCE), 2015 IEEE 78–84. doi:10.1109/ECCE.2015.7309672

Kostecki, M., 1998. The durable use of consumer products: new options for business and consumption. Kluwer Academic Publishers, Dordrecht, Netherlands.

Kota, S., Chakrabarti, A., 2007. Use of DFE methodologies and tools - major barriers and challenges, in: International Conference on Engineering Design, ICED'07 - 28-31/08/2007. Paris, France.

Krämer, S., Engell, S., 2017. Resource Efficiency of Processing Plants: Monitoring and Improvement. Wiley.

Kushnir, D., 2015. Lithium Ion Battery Recycling Technology 2015: Current State and Future Prospects. Environ. Syst. Anal. Chalmers Univ. Göteborg, Sweden. ESA Rep. # 201518 1–56.

Lacey, G., Putrus, G., Salim, A., 2013. The use of second life electric vehicle batteries for grid support. IEEE EuroCon 2013 1255–1261. doi:10.1109/EUROCON.2013.6625141

Langkau, S., Tercero Espinoza, L.A., 2018. Technological change and metal demand over time: What can we learn from the past? Sustain. Mater. Technol. 16, 54–59. doi:10.1016/J.SUSMAT.2018.02.001

Latunussa, C.E.L., Mancini, L., Blengini, G.A., Ardente, F., Pennington, D., 2016. Analysis of material recovery from photovoltaic panels. Life Cycle Assessment and Implications for Critical Raw Materials and Ecodesign. JRC-EC (Joint Research Centre - European Commission). Available at https://bookshop.europa.eu/en/analysis-of-material-recovery-from-photovoltaic-panels-pbLBNA27797/. doi:10.2788/786252

Lebedeva, N., Persio, F. Di, Boon-brett, L., 2016. Lithium ion battery value chain and related opportunities for Europe. EUR 28534 EN, Publications Office of the European Union, Luxembourg. doi:10.2760/6060, JRC105010

Lee, P., 2015. Top Five Best Vacuum Cleaners for 2014 [WWW Document]. URL http://executivecatering.hubpages.com/hub/Top-Five-Vacuum-Cleaners

Lettieri, P., Evangelisti, S., Tagliaferri, C., Domenech, T., Dijk, M., 2015. POLFREE D3.6: Report about Modelling Results from LCA Perspective. Policy Options for a Resource-Efficient Economy (POLFREE), FP7 project.

Maclean, H.L., Lave, L.B., 2003. Life Cycle Assessment of Automobile / Fuel Options. Environ. Sci. Technol. 37, 5445–5452. doi:10.1021/es034574q

Majeau-Bettez, G., Hawkins, T.R., Strømman, A.H., 2011. Life Cycle Environmental Assessment of Li-ion and Nickel Metal Hydride Batteries for Plug-in Hybrid and Battery Electric Vehicles. Environ. Sci. Technol. 45, 4548–4554. doi:10.1021/es103607c

Mancini, L., Benini, L., Sala, S., 2015. Resource footprint of Europe: Complementarity of material flow analysis and life cycle assessment for policy support. Environ. Sci. Policy 54, 367–376.

doi:10.1016/J.ENVSCI.2015.07.025

Mancini, M.R., Frangini, S., Alvani, C., 2013. Recupero eco-sostenibile di materiali da batterie litio-ione esauste: studi preliminari. Agenzia nazionale per le nuove tecnologie, l'energia e lo sviluppo economico sostenibile (ENEA). Report RdS/2013/256 Agenzia nazionale per le nuove tecnologie, l'energia e lo sviluppo economico sostenibile Ministero dello sviluppo economico.

Mathews, A., Curiel, C., Ploumis, P., 2015a. Europe scrap metal prices [WWW Document]. URL http://www.scrapmonster.com/european-scrap-prices (accessed 12.1.15).

Mathews, A., Curiel, C., Ploumis, P., 2015b. New Zinc Die Cast Scrap Price [WWW Document]. URL http://www.scrapmonster.com/scrap-metal-prices/zinc-scrap/new-zinc-die-cast/149 (accessed 5.20.12).

Matheys, J., Van Autenboer, W., Timmermans, J.-M., Van Mierlo, J., den Bossche, P., Maggetto, G., 2007. Influence of functional unit on the life cycle assessment of traction batteries. Int. J. Life Cycle Assess. 12, 191. doi:10.1065/lca2007.04.322

Mathieux, F., Ardente, F., Bobba, S., Nuss, P., Blengini, G.A., Alves Dias, P., Blagoeva, D., Torres De Matos, C., Wittmer, D., Pavel, C., Hamor, T., Saveyn, H., Gawlik, B., Orveillon, G., Huygens, D., Garbarino, E., Tzimas, E., Bouraoui, F., Solar, S., 2017. Critical raw materials and the circular economy - Background report. JRC-EC (Joint Research Centre - European Commission) Science-for-policy report, EUR 28832 EN, Publications Office of the European Union, Luxembourg. doi:10.2760/378123

Mathieux, F., Brissaud, D., 2010. End-of-life product-specific material flow analysis. Application to aluminum coming from end-of-life commercial vehicles in Europe. Resour. Conserv. Recycl. 55, 92–105. doi:10.1016/j.resconrec.2010.07.006

Mayer, A., Haas, W., Wiedenhofer, D., Krausmann, F., Nuss, P., Blengini, G.A., 2018. Measuring Progress towards a Circular Economy: A Monitoring Framework for Economy-wide Material Loop Closing in the EU28. J. Ind. Ecol. doi:10.1111/jiec.12809

McEachern, A., 2012. Hybrids: What is the Difference Between Traditional and Plug-in? [WWW Document]. Electr. Veh. News. URL http://www.fleetcarma.com/hybrids-what-is-the-difference-between-traditional-and-plug-in/ (accessed 10.28.16).

Messagie, M., Boureima, F.S., Coosemans, T., Macharis, C., Mierlo, J. Van, 2014a. A range-based vehicle life cycle assessment incorporating variability in the environmental assessment of different vehicle technologies and fuels. Energies 7, 1467–1482. doi:10.3390/en7031467

Messagie, M., Boureima, F.S., Coosemans, T., Macharis, C., Mierlo, J. Van, 2014b. A range-based vehicle life cycle assessment incorporating variability in the environmental assessment of different vehicle technologies and fuels. Energies 7, 1467–1482. doi:10.3390/en7031467

Messagie, M., Mertens, J., Oliveira, L., Rangaraju, S., Sanfelix, J., Coosemans, T., Van Mierlo, J., Macharis, C., 2014c. The hourly life cycle carbon footprint of electricity generation in Belgium, bringing a temporal resolution in life cycle assessment. Appl. Energy 134, 469–476. doi:10.1016/j.apenergy.2014.08.071

Mirae Asset Daewoo Co. Ltd., 2017. Electric Vehicles. Evolution toward third-generation EVs Evolution.

Molenbroek, E., Edith Molenbroek, Smith, M., Groenenberg, H., Waide, P., Attali, S., Fischer, C., Krivošik, J., Fonseca, P., Santos, B., Fong, J., 2014. Evaluation of Energy Labelling Directive and specific aspects of the Ecodesign Directive.

Morris, C., 2016a. Nissan and Eaton introduce home energy storage system with second-life EV batteries [WWW Document]. URL https://chargedevs.com/newswire/nissan-and-eaton-introduce-home-energy-storage-system-with-second-life-ev-batteries/ (accessed 11.1.16).

Morris, C., 2016b. Renault and Connected Energy collaborate on E-STOR energy storage product [WWW Document]. URL https://chargedevs.com/newswire/renault-and-connected-energy-collaborate-on-e-stor-energy-storage-product/ (accessed 11.1.16).

Morris, C., 2016c. BMW demonstrates stationary storage system using i3 battery [WWW Document]. URL https://chargedevs.com/newswire/bmw-demonstrates-stationary-storage-system-using-i3-battery/ (accessed 11.1.16).

Morris, C., 2015a. Daimler uses smart EV battery packs in German electricity balancing market [WWW Document]. URL https://chargedevs.com/newswire/daimler-uses-smart-ev-battery-packs-in-german-electricity-balancing-market/ (accessed 11.1.16).

Morris, C., 2015b. Portable charging station powered by second-life EV batteries [WWW Document]. URL https://chargedevs.com/newswire/portable-charging-station-powered-by-second-life-ev-batteries (accessed 11.1.16).

Mudgal, S., 2010. Approaches to eco-innovation for SMEs regarding energy using/related products, in: Making Eco-Innovation Happen in Small and Medium-Sized Enterprises. 8th European Forum on Eco-Innovation, Bilbao, Spain 20-21 April 2010.

Mudgal, S., Benito, P., 2008. Reporting on the Implementation of Integrated Product Policy (IPP). BIO Intelligence Service, prepared for the European Commission - DG ENV.

Mudgal, S., Kong, M.A., Mitsios, A., Pahal, S., Lecerf, L., Acoleyen, M. Van, Lambert, S., Fedrigo, D., Watkins, E., 2014. Ex-post evaluation of certain waste stream Directives. Final report. Bio Intelligence Service prepared for the European Commission - DG ENV. Available at http://ec.europa.eu/environment/waste/pdf/target_review/Final%20Report%20Ex-Post.pdf.

Müller, E., Hilty, L.M., Widmer, R., Schluep, M., Faulstich, M., 2014. Modeling metal stocks and flows: A review of dynamic material flow analysis methods. Environ. Sci. Technol. 48, 2102–2113. doi:10.1021/es403506a

Murakami, S., Oguchi, M., Tasaki, T., Daigo, I., Hashimoto, S., 2010. Lifespan of Commodities, Part I. J. Ind. Ecol. 14, 598–612. doi:10.1111/j.1530-9290.2010.00250.x

Nakamura, S., Kondo, Y., Kagawa, S., Matsubae, K., Nakajima, K., Nagasaka, T., 2014. MaTrace: Tracing the fate of materials over time and across products in open-loop recycling. Environ. Sci. Technol. 48, 7207–7214. doi:10.1021/es500820h

Narula, C.K., Martinez, R., Onar, O., Starke, M.R., Andrews, G., 2011. Economic Analysis of Deploying Used Batteries in Power Systems. ORNL/TM-2011/151. Oak Ridge National laboratory - US Department of Energy. Oak Ridge (Tennessee).

Natkunarajah, N., Scharf, M., Scharf, P., 2015. Scenarios for the return of lithium-ion batteries out of electric cars for recycling. Procedia CIRP 29, 740–745. doi:10.1016/j.procir.2015.02.170

Nelson, P.A., Gallagher, K.G., Bloom, I., Dees, D.W., 2011. Modeling the Performance and Cost of Lithium-Ion Batteries for Electric-Drive Vehicles. ARGONNE National laboratory (ANL). doi:10.2172/1027714

Nemry, F., Leduc, G., Mongelli, I., Uihlein, A., 2008. Environmental Improvement of Passenger Cars (IMPRO-car). European Commission, Joint Research Centre (JRC), Institute for Prospective Technological Studies (ipts). doi:10.2791/63451

Nenadic, N.G., Bussey, H.E., Ardis, P.A., Thurston, M.G., Letters, T., 2014. Estimation of State-of-Charge and Capacity of Used Lithium-Ion Cells. Int. J. Progn. Heal. Manag. 1–12.

NETfficient - Storage for Life. Available at http://netfficient-project.eu/ (accessed 11.1.16).

Neubauer, J., Pesaran, A., 2011. The ability of battery second use strategies to impact plug-in electric vehicle prices and serve utility energy storage applications. J. Power Sources 196, 10351–10358. doi:10.1016/j.jpowsour.2011.06.053

Neubauer, J., Smith, K., Wood, E., Pesaran, A., 2015a. Identifying and Overcoming Critical Barriers to Widespread Second Use of PEV Batteries. National Renewable Energy Laboratory (NREL). Available at http://www.nrel.gov/docs/fy15osti/63332.pdf.

Neubauer, J., Wood, E., Pesaran, A., 2015b. A Second Life for Electric Vehicle Batteries: Answering Questions on Battery Degradation and Value. SAE Int. J. Mater. Manuf. 8, 21–23. doi:10.4271/2015-01-1306

Nicholson, A., 2009. Methods for managing uncertainty in material selection decisions: Robustness of early stage Life Cycle Assessment. Dep. Mech. Eng.

Nordelöf, A., Messagie, M., Tillman, A.M., Ljunggren Söderman, M., Van Mierlo, J., 2014. Environmental impacts of hybrid, plug-in hybrid, and battery electric vehicles-what can we learn from life cycle assessment? Int. J. Life Cycle Assess. 19, 1866–1890. doi:10.1007/s11367-014-0788-0

Notter, D.A., Gauch, M., Widmer, R., Wäger, P., Stamp, A., Zah, R., Althaus, H.J., 2010. Contribution of Li-ion batteries to the environmental impact of electric vehicles. Environ. Sci. Technol. 44, 6550–6556. doi:10.1021/es903729a

NREL (National Renewable Energy Laboratory), 2015. Battery Second Use for Plug-In Electric Vehicles [WWW Document]. URL http://www.nrel.gov/transportation/energystorage/use.html (accessed 11.1.16).

Nuss, P., Blengini, G.A., 2018. Towards better monitoring of technology critical elements in Europe: Coupling of natural and anthropogenic cycles. Sci. Total Environ. 613–614, 569–578. doi:10.1016/J.SCITOTENV.2017.09.117

Oeko Institute, 2017. Assessment of the implementation of Directive 2000/53/EU on end-of-life vehicles (the ELV Directive) with emphasis on the end of life vehicles of unknown whereabouts. Oko Institute e.V. – Institute for Applied Ecology. Darmstadt (Germany).

Oers, L. van, A. de Koning, J.B. Guinée & G. Huppes, 2002. Abiotic resource depletion in LCA - Improving characterisation factors for abiotic resource depletion as recommended in the new Dutch LCA Handbook. DWW report, Delft.

Oko Institute, 2016. Assessment of the implementation of Directive 2000/53 /EC on end-of life vehicles (the ELV Directive) with emphasis on the end-of life vehicles of unknown whereabouts. Oko Institute e.V. – Institute for Applied Ecology. Darmstadt (Germany).

Okumura, S., Morikuni, T., Okino, N., 2001. Life design for remanufacturing durable products. Proc. Second Int. Symp. Environ. Conscious Des. Inverse Manuf. doi:10.1109/.2001.992364

Oliveira, L.M., 2017. Environmental aspects of lithium-ion energy storage systems using a multi-stage hybrid approach combining the benefits of life cycle assessment and input-output models. Electrical Engineering and Energy Technology - VUB University (Brussels).

Olivetti, E.A., Ceder, G., Gaustad, G.G., Fu, X., 2017. Lithium-Ion Battery Supply Chain Considerations: Analysis of Potential Bottlenecks in Critical Metals. Joule 1, 229–243. doi:10.1016/j.joule.2017.08.019

Olivetti, E.A., Duan, H., Kirchain, R.E., 2012. Exploration into the environmental assessment of electrical products available at http://www.nema.org/Policy/Environmen.

Ordoñez, J., Gago, E.J., Girard, A., 2016. Processes and technologies for the recycling and recovery of spent lithium-ion batteries. Renew. Sustain. Energy Rev. 60, 195–205. doi:10.1016/J.RSER.2015.12.363

Parker, D., Riley, K., Robinson, S., Symington, H., Hollins, O., 2015. Remanufacturing Market Study, European Remanufacturing Network.

Patry, G., Romagny, A., Froelich, D., Patry, G., Romagny, A., 2014. Cost modeling of lithium-ion battery cells for automotive applications. Energy Sci. Eng. doi:10.1002/ese3.47

Pavel, C.C., Blagoeva, D.T., 2017. Materials impact on the EU 's competitiveness of the renewable energy, storage and e-mobility sectors. doi:10.2760/83521

Payet, J. (2006)., 2006. Deliverable D.4.1.4 Report describing a method for the quantification of impacts on aquatic freshwater ecosystems resulting from different stressors (e.g., toxic substances, eutrophication, etc).

PE Europe GMBH, 2011. Professional database. Software GaBi, ver. 6 [WWW Document]. URL www.gabi-software.com

Pehlken, A., Albach, S., Vogt, T., 2017. Is there a resource constraint related to lithium ion batteries in cars? Int. J. Life Cycle Assess. 22, 40–53. doi:10.1007/s11367-015-0925-4

Pelletier, N., Ardente, F., Brandão, M., De Camillis, C., Pennington, D., 2015. Rationales for and limitations of preferred solutions for multi-functionality problems in LCA: is increased consistency possible? Int. J. Life Cycle Assess. 20, 74–86. doi:10.1007/s11367-014-0812-4

Perks, C., 2016. The role of cobalt in battery supply [WWW Document]. URL http://www.indmin.com/Article/3586106/The-role-of-cobalt-in-battery-supply.html (accessed 8.20.08).

Peters, J.F., Baumann, M., Zimmermann, B., Braun, J., Weil, M., 2017. The environmental impact of Li-Ion batteries and the role of key parameters – A review. Renew. Sustain. Energy Rev. 67, 491–506. doi:10.1016/j.rser.2016.08.039

Peters, J.F., Weil, M., 2018. Providing a common base for life cycle assessments of Li-Ion batteries. J. Clean. Prod. 171, 704–713. doi:10.1016/J.JCLEPRO.2017.10.016

Petersen, J., 2018. How Battery Chemistry Assumptions Distort Nickel And Cobalt Demand Forecasts [WWW Document]. URL https://seekingalpha.com/article/4152035-battery-chemistry-assumptions-distort-nickel-cobalt-demand-forecasts?page=3 (accessed 9.6.18).

Pillot, C., 2017. The Rechargeable Battery Market and Main Trends 2016-2025. Adv. Automot. Batter. Conf.

Planet Ark, 2007. Prospecting the Future of Recycling. Sydney NSW.

Podias, A., Pfrang, A., Di Persio, F., Kriston, A., Bobba, S., Mathieux, F., Messagie, M., Boon-Brett, L., 2018. Sustainability Assessment of Second Use Applications of Automotive Batteries: Ageing of Li-Ion Battery Cells in Automotive and Grid-Scale Applications. World Electr. Veh. J. 9, 24. doi:10.3390/wevj9020024

Posch, M., Seppälä, J., Hettelingh, J.P., Johansson, M., Margni M., Jolliet, O., 2008. The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA.

Pourmohammadi, H., Rahimi, M., Dessouky, M., 2008. Sustainable reverse logistics for distribution of industrial waste/byproducts: A joint optimization of operation and environmental costs. Supply Chain Forum An Int. J. 9, 2–17.

Ramirez, D.A.B., Ochoa, G.E.V., Peña, A.R., Escorcia, Y.C., 2018. Bibliometric analysis of nearly a decade of research in electric vehicles: a dynamic approach. ARPN J. Eng. Appl. Sci. 13, 4730–4736.

Recharge, 2014. Re-use and Second use of Rechargeable Batteries. Advanced Rechargeable Batteries (RECHARGE) Association. Available at http://www.rechargebatteries.org/wp-content/uploads/2014/04/RECHARGE-Information-Paper-on-Re-use-and-second-use-October-2014-v.14.pdf.

Recharge Association, 2018. PEFCR - Product Environmental Footprint Category Rules for High

Specific Energy Rechargeable Batteries for Mobile Applications.

Rehme, M., Richter, S., Temmler, A., Götze, U., 2016. Second-Life Battery Applications - Market potentials and contribution to the cost effectiveness of electric vehicles, in: 5th Conference on Future Automotive Technology (CoFAT). Munich (May 2009). doi:10.13140/RG.2.2.35238.93760

Reinhardt, R., Domingo, S.G., Garcia, B.A., Christodoulou, I., 2017. Macro environmental analysis of the electric vehicle battery second use market. Int. Conf. Eur. Energy Mark. EEM. doi:10.1109/EEM.2017.7982031

ReMaTec, 2018. Batteries not included [WWW Document]. URL https://www.rematec.com/news-articles/batteries-not-included/ (accessed 2.2.19).

Research, M., 2016. Lithium market outlook Charging up , but only limited range.

Richa, K., 2016. Sustainable management of lithium-ion batteries after use in electric vehicles.

Richa, K., Babbitt, C.W., Gaustad, G., 2017. Eco-Efficiency Analysis of a Lithium-Ion Battery Waste Hierarchy Inspired by Circular Economy. J. Ind. Ecol. 21, 715–730. doi:10.1111/jiec.12607

Richa, K., Babbitt, C.W., Gaustad, G., Wang, X., 2014. A future perspective on lithium-ion battery waste flows from electric vehicles. Resour. Conserv. Recycl. 83, 63–76. doi:10.1016/j.resconrec.2013.11.008

Richa, K., Babbitt, C.W., Nenadic, N.G., Gaustad, G., 2015. Environmental trade-offs across cascading lithium-ion battery life cycles. Int. J. Life Cycle Assess. 20, 1–16. doi:10.1007/s11367-015-0942-3

RIM, 2015. Shredded Aluminum - International Scrap Metal Prices [WWW Document]. URL http://www.recycleinme.com/scrapresources/detailedprice.aspx?psect=1&cat=International Scrap Metal Prices&subcat=Shredded Aluminum&from=01/1/2015&to=11/27/2015 (accessed 5.20.12).

Roghanian, E., Pazhoheshfar, P., 2014. An optimization model for reverse logistics network under stochastic environment by using genetic algorithm. J. Manuf. Syst. 33, 348–356. doi:10.1016/j.jmsy.2014.02.007

Rohr, S., Müller, S., Baumann, M., Kerler, M., Ebert, F., Kaden, D., Lienkamp, M., 2017a. Quantifying Uncertainties in Reusing Lithium-Ion Batteries from Electric Vehicles. Procedia Manuf. 8, 603–610. doi:10.1016/j.promfg.2017.02.077

Rohr, S., Wagner, S., Baumann, M., Muller, S., Lienkamp, M., 2017b. A techno-economic analysis of end of life value chains for lithium-ion batteries from electric vehicles. 2017 Twelfth Int. Conf. Ecol. Veh. Renew. Energies 1–14. doi:10.1109/EVER.2017.7935867

Rose, C.M., Stevels, A., 1999. Product Characteristics and their Relationship to End-of-Life Strategies, in: Proceedings of the 1998 IEEE International Symposium, Electronics and the Environment.

Rotter, V.S., 2017. Material flow in a circular economy, in: ICBR - International Congress on Battery Recycling. Lisbon (Portugal).

Ruiz, V., Boon-Brett, L., Steen, M., Berghe, L. Van den, 2016. Putting Science into Standards: Workshop - Summary & Outcomes. Driving Towards Decarbonisation of Transport: Safety, Performance, Second life and Recycling of Automotive Batteries for e-Vehicles. Workshop organised by JRC and CEN-CENELEC JRC Petten (The Netherlands), 22-23 September 2016.

Ruoff, C., 2016. Second life: Spiers New Technologies develops advanced battery classification techniques [WWW Document]. URL https://chargedevs.com/features/second-life-spiers-new-technologies-develops-advanced-battery-classification-techniques/ (accessed 12.1.16).

Sabbaghi, M., Cade, W., Behdad, S., Bisantz, A.M., 2017. The current status of the consumer electronics repair industry in the U.S.: A survey-based study. Resour. Conserv. Recycl. 116, 137–151. doi:10.1016/J.RESCONREC.2016.09.013

Saez-de-Ibarra, A., Martinez-Laserna, E., Koch-Ciobotaru, C., Rodriguez, P., Stroe, D.-I., Swierczynski, M., 2015. Second life battery energy storage system for residential demand response service. 2015 IEEE Int. Conf. Ind. Technol. 2941–2948. doi:10.1109/ICIT.2015.7125532

Sala, S., Pant, R., Hauschild, M., Pennington, D., 2012. Research needs and challenges from science to decision support. Lesson learnt from the development of the international reference life cycle data system (ILCD) recommendations for life cycle impact assessment. Sustainability 4, 1412–1425. doi:10.3390/su4071412

Sathre, R., Scown, C.D., Kavvada, O., Hendrickson, T.P., 2015. Energy and climate effects of second-life use of electric vehicle batteries in California through 2050. J. Power Sources 288, 82–91. doi:10.1016/j.jpowsour.2015.04.097

Schmitt, G., Scott, J., Davis, A., Utz, T., 2016. Patents and progress; intellectual property showing the future of electric vehicles. World Electr. Veh. J. 8, 635–645.

Schoenwald, D., Ellison, J., 2016. Determination of Duty Cycle for Energy Storage Systems in a Renewables (Solar) Firming Application, SAND2016-3636. Sandia National Laboratories, Albuquerque, New Mexico 87185 and Livermore, California 94550 (USA).

Schultmann, F., Engels, B., Rentz, O., 2003. Closed-Loop supply chain for spent batteries. Interfaces (Providence). 33, 57–71. doi:10.1287/inte.33.6.57.25183

Seppälä, J., Posch, M., Johansson, M., Hettelingh, J.P., 2006. Country-dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator.

Shibasaki, M., Warburg, N., Eyerer, P., 2005. Directives and legislations - Recycling and reuse of products, in: Proceedings - Fourth International Symposium on Environmentally Conscious Design and Inverse Manufacturing, Eco Design 2005. doi:10.1109/ECODIM.2005.1619358

Simon, B., Ziemann, S., Weil, M., 2015. Potential metal requirement of active materials in lithium-ion battery cells of electric vehicles and its impact on reserves: Focus on Europe. Resour. Conserv. Recycl. 104, 300–310. doi:10.1016/j.resconrec.2015.07.011

Skinner, I., van Essen, H., Smokers, R., Hill, N., 2010. Towards the decarbonisation of EU's transport sector by 2050. Final report of the EU References 74 Electric vehicles from life cycle and circular economy perspectives Routes to 2050 project for the European Commission (DG CLIMA).

Sneck, T., 1981. RILEM and durability. Mater. Struct. 14, 379–390.

Spielmann, M., Althaus, H.J., 2007. Can a prolonged use of a passenger car reduce environmental burdens? Life Cycle analysis of Swiss passenger cars. J. Clean. Prod. 15, 1122–1134. doi:10.1016/j.jclepro.2006.07.022

Stahel, W.R., 2013. Policy for material efficiency--sustainable taxation as a departure from the throwaway society. Philos. Trans. R. Soc. A Math. Phys. Eng. Sci. 371. doi:10.1098/rsta.2011.0567

Stahl, H., Baron, Y., Hay, D., Hermann, A., Mehlhart, G., Baroni, L., Rademaekers, K., Williams, R., Pahal, S., 2018. Study in support of evaluation of the Directive 2006/66/EC on batteries and accumulators and waste batteries and accumulators. Final Report.

Standridge, C.R., Corneal, L., Baine, N., 2016. Advances in Repurposing and Recycling of Post-Vehicle-Application Lithium-Ion Batteries. Mineta National Transit Research Consortium (MNTRC). REPORT 12-55. San José State University (CA).

Steen, B., 1999a. A Systematic approach to environmental priority strategies in product development (EPS). Version 2000-Models and data of the default method; CPM report 1999:5.

Steen, B., 1999b. Environmental Assessment of Products and Material Systems A systematic approach to environmental priority strategies in product development ( EPS ). Version 2000 – General system characteristics 66.

Stoner, S., Olson, T., Addy, M., Tuvell, T., Shapiro, R., 2007a. Full Fuel Cycle Assessment Well to Wheels Energy Inputs, Emissions, and Water Impacts. TIAX, LCC for California Energy Commission.

Stoner, S., Olson, T., Addy, M., Tuvell, T., Shapiro, R., 2007b. Full Fuel Cycle Assessment Well to Tank Energy Inputs, Emissions, and Water Impacts.

Stringer, D., Ma, J., 2018. Where 3 Million Electric Vehicle Batteries Will Go When They Retire [WWW Document]. URL https://www.bloomberg.com/news/features/2018-06-27/where-3-million-electric-vehicle-batteries-will-go-when-they-retire

Sumitomo, 2014. Used Nissan Leaf batteries support solar energy system in Japan [WWW Document]. URL http://www.autoblog.com/2014/02/18/used-nissan-leaf-batteries-support-solar-energy-system-japan/. (accessed 11.1.16).

Swain, B., 2017. Recovery and recycling of lithium: A review. Sep. Purif. Technol. 172, 388–403. doi:10.1016/j.seppur.2016.08.031

Sweeting, W.J., Winfield, P.H., 2012. Future transportation: Lifetime considerations and framework for sustainability assessment. Energy Policy 51, 927–938. doi:10.1016/J.ENPOL.2012.09.055

Tamiang, A., Angka, D., 2014. Electrode and cells manufacturing white paper - chapter 5: Batteries and end of life management. European Li-ion battery advanced manufacturing for electric vehicles (ELIBAMA). Available at: https://elibama.wordpress.com/electrodes-and-cells-manufacturing-white-paper/.

Tasaki, T., Motoshita, M., Uchida, H., Suzuki, Y., 2013. Assessing the Replacement of Electrical Home Appliances for the Environment: An Aid to Consumer Decision Making Tasaki et al. Replacement of Home Appliances for the Environment. J. Ind. Ecol. 17, 290–298. doi:10.1111/j.1530-9290.2012.00551.x

Taylor, N., Szabo, S., Kona, A., Melica, G., Huld, T., Ossenbrink, H., Commission, E., 2015. Deployment pathways for photovoltaics in the EU towards 2020: comparing economic factors with policies at municipal level. Eu Pvsec 2015 1–8.

Tecchio, P., Fabio, M.D.S., Ardente, F., Pekar, F., Mathieux, F., 2019. Sustainable use of Materials through Automotive Remanufacturing to boost resource efficiency in the road Transport system. JRC internat report.

Technologies, S.N., 2015. Lifecycle Management of Advanced Vehicle Batteries and Beyond. Available at http://www.epri.com/Documents/ENV-Vision/EMERGING AND CROSS-CUTTING ISSUES/Environmental Considerations of Grid Scale Battery Storage/Dirk Spiers_Spiers New Technologies.pdf.

The, Q., Council, Q., Affairs, E.. EU Sustainable Consumption and Production Action Plan The Quaker Council for European Affairs.

Thomas, M., Ellingsen, L.A.-W., Hung, C.R., 2018. Research for TRAN Committee - Battery-powered electric vehicles: market development and lifecycle emissions. Study requested by the European Parliament's Committee on Transport and Tourism. Available at http://www.europarl.europa.eu/thinktank/en/document.html?reference=IPOL_STU(2018)617457.

Timmers, H., 2016. Driving Towards Decarbonisation of Transport: Safety, Performance, Second life and Recycling of Automotive Batteries for e-Vehicles - Recycling, in: Putting Science into Standards (PSIS) Workshop 22-23/09/2016. Petten (The Netherlands).

Tivander, J., 2016. Environmental concerns of metals in Li-ion batteries Implications for recycling. Department of Energy and Environment of the Chalmers University of Technology (Sweden).

TNS, 2014. Flash Eurobarometer 388. Attitudes of Europeans towards resource efficiency, Flash Eurobarometer No 316. TNS Political & Social. doi:10.2779/1 482

Tong, S.J., Same, A., Kootstra, M.A., Park, J.W., 2013. Off-grid photovoltaic vehicle charge using second life lithium batteries: An experimental and numerical investigation. Appl. Energy 104, 740–750. doi:10.1016/j.apenergy.2012.11.046

Tukker, A., Huppes, G., Guinée, J., Heijungs, R., de Koning, A., van Oers, L., Suh, S., Geerken, T., Van Holderbeke, M., Jansen, B., Nielsen, P., 2006. Environmental Impact of Products (EIPRO). Analysis of the life cycle environmental impacts related to the final consumption of the EU-25. European Commission, Joint Research Centre (JRC, Institute for Prospective Technological Studies (ipts).

van Nes, N., Cramer, J., 2006. Product lifetime optimization: a challenging strategy towards more sustainable consumption patterns. J. Clean. Prod. 14, 1307–1318. doi:10.1016/j.jclepro.2005.04.006

Van Zelm, R., Huijbregts, M.A.J., Den Hollander, H.A., Van Jaarsveld, H.A., S., F.J., Struijs, J., Van Wijnen, H.J., Van de Meent, D., 2008. European characterization factors for human health damage of PM10 and ozone in life cycle impact assessment. Atmos. Environ. 42, 441-453.

Vandenbergh, M., 2014. Solar photovoltaic production at JRC Petten – monitoring report. JRC-EC (Joint Research Centre - European Commission) technical report (JRC93782).

Vieceli, N., Nogueira, C.A., Guimarães, C., Pereira, M.F.C., Durão, F.O., Margarido, F., 2018. Hydrometallurgical recycling of lithium-ion batteries by reductive leaching with sodium metabisulphite. Waste Manag. 71, 350–361. doi:10.1016/J.WASMAN.2017.09.032

Viswanathan, V. V., Kintner-Meyer, M., 2011. Second use of transportation batteries: Maximizing the value of batteries for transportation and grid services. IEEE Trans. Veh. Technol. 60, 2963–2970. doi:10.1109/TVT.2011.2160378

Weidema, B.P., Norris, G. a, 2002. Avoiding co-product allocation in the metals sector. ICMM Int. Work. Life Cycle Assess. Met.

Weniger, J., Bergner, J., Quaschning, V., 2014a. Integration of PV power and load forecasts into the operation of residential PV battery systems. 4th Sol. Integr. Work. 383–390. doi:10.13140/2.1.3048.9283

Weniger, J., Tjaden, T., Quaschning, V., 2014b. Sizing of residential PV battery systems. Energy Procedia 46, 78–87. doi:10.1016/j.egypro.2014.01.160

Williams, B., 2011. Plug-In-Vehicle Battery Second Life: the Effect of Post-Vehicle, Distributed-Grid Energy Storage Value on Battery-Lease Payments.

Williams, B., Lipman, T., 2011. Analysis of the Combined Vehicle - Use Value of Lithium-Ion Plug-In-Vehicle Propulsion Batteries - Task 3, Second Life Applications and Value of “Traction” Lithium Batteries. Prepared for the California Energy Commission (CEC). University of California, Berkeley. Transportation Sustainability Research Center.

Williams, I.D., Shaw, P.J., 2017. Reuse: Fashion or future? Waste Manag. 60, 1–2. doi:10.1016/J.WASMAN.2017.02.017

Willson, Q., 2018. Why it Makes Sense to Buy a Used EV [WWW Document]. URL http://www.routemonkey.com/drivers/everything-ev/why-it-makes-sense-to-buy-a-used-ev

Winkler, J., Regawitz, M., 2016. Solar energy policy in the EU and the Member States, from the perspective of the petitions received. Fraunhofer Institute for Systems and Innovation Research (ISI) for the European Parliament's Committee on Petitions and commissioned, supervised and published by the Policy Department for Citizens' Rights and Constitutional Affairs. Available at http://ww. doi:10.1016/B978-0-12-809806-6.00015-8

Wolf, M.A., Pant, R., Chomkhamsri, K., Sala, S., Pennington, D., 2012. The International Reference Life Cycle Data System (ILCD) Handbook (online version). doi:10.2788/85727

Wollerton, M., 2013. Vacuum buying guide [WWW Document]. URL http://www.cnet.com/topics/vacuum-cleaners/buying-guide/ (accessed 12.1.15).

Wong, Y.L., 2009. An integrated environmental product assessment model on ecodesign for electrical products and electronic appliances. Department of Industrial and System Engineering of the Hong Kong Polytechnic University.

WRAP, 2010. Environmental assessment of consumer electronic products. WRAP - Waste & Resources Action Pogramme.

Yun, L., Linh, D., Shui, L., Peng, X., Garg, A., LE, M.L.P., Asghari, S., Sandoval, J., 2018. Metallurgical and mechanical methods for recycling of lithium-ion battery pack for electric vehicles. Resour. Conserv. Recycl. 136, 198–208. doi:10.1016/j.resconrec.2018.04.025

Zackrisson, M., Avellán, L., Orlenius, J., 2010. Life cycle assessment of lithium-ion batteries for plug-in hybrid electric vehicles – Critical issues. J. Clean. Prod. 18, 1519–1529. doi:10.1016/j.jclepro.2010.06.004

Zanotti, S., Scolari, F., 2012. Produzione pani di zinco [WWW Document]. URL http://www.valdelriso.it/modules/myalbum/photo.php?lid=316 (accessed 12.1.15).

Zhao, X., Wang, S., Wang, X., 2018. Characteristics and Trends of Research on New Energy Vehicle Reliability Based on the Web of Science. doi:10.3390/su10103560

Ziemann, S., Müller, D.B., Schebek, L., Weil, M., 2018. Modeling the potential impact of lithium recycling from EV batteries on lithium demand: A dynamic MFA approach. Resour. Conserv. Recycl. 133, 76–85. doi:10.1016/j.resconrec.2018.01.031

Zubi, G., Dufo-López, R., Carvalho, M., Pasaoglu, G., 2018. The lithium-ion battery: State of the art and future perspectives. Renew. Sustain. Energy Rev. 89, 292–308. doi:10.1016/J.RSER.2018.03.002

# List of Figures

# List of Tables

# Appendix A

## Goal and scope definition

## 11.1 Six aspects of the goal definition

**General info about LCA (from ILCD/ISO):**

In defining the goal of an LCA, the following items shall be unambiguously stated:

- the intended application
- the reasons for carrying out the study
- the intended audience, i.e. to whom the results of the study are intended to be communicated
- whether the results are intended to be used in comparative assertions intended to be disclosed to the public.

Further, also the commissioner of the study and the influential actors should be listed.

**LCA of car**:

The overall goal of LCA analysis applied to vehicles, and cars in detail, is the assessment of the environmental performances of vehicles in order to control and regulate the different phases of their life-cycle (i.e. manufacturing, distribution, use-phase, EoL). Furthermore, as the LCA permits to identify the most important hot-spots from an environmental perspective, this tool could also be adopted in the design phase, in order to be aware and, if possible, to prevent potential environmental burdens.

The reasons behind the LCA of cars are connected to the global recognition of the importance in creating sustainable production and consumption patterns in order to reduce the existing social and environmental impact[90]. The concept of Life Cycle Thinking (LCT), which aims at identifying possible improvements to goods and services in the form of lower environmental impacts and reduced use of resources across all life cycle stages, is becoming extremely diffused and important in all industrial sectors, reflecting in specific policies and regulations issued at European and National level, as the Sustainable Consumption and Production Action Plan (The Quaker Council for European Affairs), which aims at reducing the overall environmental impact and consumption of resources associated with the complete life cycles of goods and services and the Integrated Product Policy Communication (EC, 2003), making the reduction of environmental impacts one of the main targets of industrial and governmental policies at different levels. In this framework, the efforts focused on developing a sustainable transport system in Europe both for light-duty vehicles (cars and light vans) and heavy-duty vehicles

[90] http://www.unep.org/roe/KeyActivities/SustainableConsumptionTransport/tabid/54068/Default.aspx#horizontalTab1

(trucks and buses) are addressed through different tools: different strategies adopted at the international level (e.g. CAFE programme and the Thematic strategy on air pollution[91]), car labelling schemes[92] (Directive 1999/94/EC, 1999) and legislative constraints concerning the reduction of $CO_2$ emissions from passenger cars[93] (EU, 2014).

The intended audience will be LCA practitioners employed in light-duty vehicles (more specifically in cars) design activities. The output of this study shall not directly be used in comparative assertions or in reports disclosed to the public.

**Comment**:

Even if this report will not be disclosed to the general public, excerpts of it and/or an extended abstract can be disclosed in the future through FCA website.

## 11.2 Classifying the decision-context

**General info about LCA (from ILCD/ISO):**

ILCD handbook differentiates between three archetypal goal situations describing possible study cases encountered in LCA.

According to Figure 1, the situation A and B are relevant when the LCA study will be used as a decision support tool on micro and macro level scale respectively, whereas situation C relates to a purely descriptive documentation of the system under analysis, without being interested in any potential consequences on other parts of the economy.

Figure 88: Archetypal goal situations (EC - JRC 2012a)

| | | | |
|---|---|---|---|
| Decision support? | | Kind of process-changes in background system / other systems | |
| | | None or small-scale | Large-scale |
| | Yes | Situation A<br>"Micro-level decision support" | Situation B<br>"Meso/macro-level decision support" |
| | No | Situation C<br>"Accounting"<br>(with C1: including interactions with other systems, C2: excluding interactions with other systems) | |

**LCA of car**:

[91] http://ec.europa.eu/environment/archives/cafe/index.htm

[92] http://ec.europa.eu/clima/policies/transport/vehicles/labelling/index_en.htm

[93] http://ec.europa.eu/clima/policies/transport/vehicles/cars/index_en.htm

In this case, the main interest is addressed in using LCA outcomes as a decision making support tool to identify the environmental hot-spots and analyse different potential scenarios concerning several aspects and/or components of the life-cycle of cars.

For these reasons, the archetypal goal situation that more represents our case study is situation A, as it refers to product-related questions. Thus, LCA could be useful in detecting the Key Environmental Performance Indicators (KEPI) for eco-design or benchmarking, weak point analysis of a product/component, eco-design/design-for-recycling, ecolabel criteria, greening the supply chain, etc. Moreover, information associated to the LCA may be used in order to support decisions within industries but also for consumers.

**Comment:**

Cars market is strongly influenced by the environmental performances of its products. Users are paying always more attention to vehicle emissions as they low emission is considered an added value of a car. Therefore, the LCT approach should be used both for taking important decision about specific components/feature of cars during their life-cycle stages, and for having reliable and complete information for consumers.

## 11.3 Function, functional unit and reference flow

**General info about LCA (from ILCD/ISO):**

The functional unit (FU) defines the quantification of the identified function(s) of the product. Primary purpose of a functional unit is to provide a reference to which the inputs and outputs are related.

Reference flow is the flow (or flows in case of multifunctional processes) to which all other input and output flows quantitatively relate (i.e. the amount of products needed to fulfil the provided function).

**LCA of car:**

Depending on the goal of the study this can be either a function of a single component (e.g. motor, transmission, etc), a life cycle stage (e.g. manufacturing, use phase, EoL) or of the whole vehicle (e.g. a specific model).

In order to assess the environmental performances of vehicles, the most likely option is to consider the car along its whole life cycle. More in-depth analysis could be done for specific life-cycle stages but the importance of performing a complete life-cycle is underlined by the international literature about LCA applied to passenger vehicles (Chlopek and Lasocki, 2011), (Messagie et al., 2014a), Nemry et al. 2008, Molenbroek et al. 2014).

It is worthy that different functional unit may be chosen depending on the main goal of the study and the function of the of the system. Among the LCA studies about passenger cars, some of the employed functional unit are: 1 car, 1 passenger trip, 1 travelled km, 1 passenger km travelled (PKT).

Though this choice, a detailed analysis of functional unit environmental impacts could be performed and the phases and single processes contributing substantially to the impacts (hot spot and contribution analysis) could be identified.

Moreover, the choice of a determined functional unit won't permit the comparison between different LCA studies with different functions (even though with the same topic: light-duty vehicles).

**Comment:**

According to Directive 2009/33/CE , the functional unit is considered in base on a mileage equal to 200.000 km during all life cycle.

Therefore, it is recommended to evaluate this type of data.

## 11.4 Inventory modelling framework

### *1.4.1 LCI modelling principle*

**General info about LCA (from ILCD/ISO):**

Two main LCI modelling principles are in use in LCA practice: attributional and consequential modelling:

- Attributional life cycle modelling depicts system's actual (or forecasted) specific or average supply-chain plus use and end-of-life value chain. The existing or forecasted system being embedded into a static technosphere.
- Consequential life cycle modelling depicts the generic supply-chain as it is theoretically expected in consequence of the analysed decision.

### *1.4.2 Multifunctionality*

**General info about LCA (from ILCD/ISO):**

If a process provides more than one function, i.e. delivering several goods and/or services (co-products), it is defined as "multifunctional". ISO 14044:2006 and ILCD present a hierarchy of different approaches to solve multifunctionality: subdivision, system expansion and allocation.

## 11.5 System boundary and cut-off criteria

### *1.5.1 System boundaries*

**General info about LCA (from ILCD/ISO):**

According to ILCD guidelines, the system boundaries define which parts of the life cycle and which processes belong to the analysed system, i.e. are required for providing its function as defined by its functional unit. They hence separate the analysed system from the rest of the technosphere. At the same time, the system boundaries also define the boundary between the analysed system and the ecosphere, i.e. define across which boundary the exchange of elementary flows with nature takes place.

**LCA of cars:**

According to the ILCD/ISO definition, the life-cycle of products include all the stages from the raw materials extraction to the EoL of the assessed product. In the automotive sector, this means that manufacturing of the car components, their assembly, the use phase as well as the EoL of the car should be assessed in order to perform a *from-cradle-to-grave* LCA.

It is worthy that, due to the importance of the use-phase of vehicles, several LCA case-studies focus just on some of the abovementioned life-cycle stages, with particular reference to the fuel cycle. Indeed, recommendations emerging from the international literature underline the importance of considering also the equipment life-cycle. Clearly, the choice of the system boundaries should be consistent with the main goal of the LCA study.

The manufacturing phase includes all the input (i.e. materials and energy) and the output (i.e. emissions, wastes, co-products, scraps, etc.) associated to the production of car components and their assembly. Therefore, extraction, transports of raw materials, components manufacturing, transports of components to the assembly plant and assembly processes should be accounted.

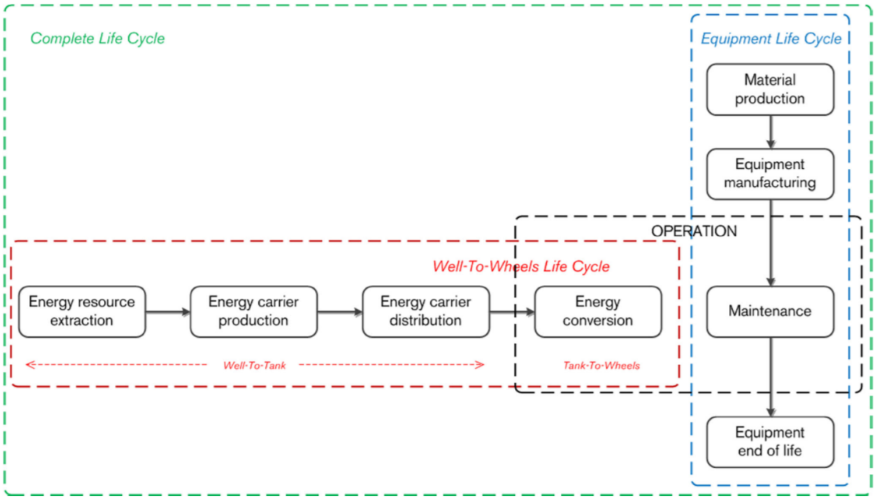
The use-phase of light-duty vehicles mostly refers to the operations permitting the usage of the product, that means the energy conversion and the maintenance of vehicles(Nordelöf et al., 2014). It is worthy that the energy conversion includes both the environmental burdens deriving from the well-to-tank and the tank-to-wheels stages. This means that all the fuel cycle is assessed: the energy resource extraction, the energy carrier production, its distribution and its conversion during the vehicle use-phase.

Finally, the EoL comprises all activities that have to be performed in order to safely dispose the vehicles and its components. Disassembly, EoL pre-treatments, recycling/recovery/reuse processes and landfill disposal should be assessed. Note that, for particular vehicles categories, the EoL could be important (particularly in terms of environmental credits) for specific impact categories. For instance, the presence of some elements (e.g. nickel, lithium, rare earths, etc.) is relevant for the resource depletion or human toxicity.

**Comment:**

Note that the performance of complete LCA permits to avoid the impacts shifting among different life-cycle stages and different environmental and human health problem fields. Moreover, the existence of several constraints (legislation, ecolabel schemes, etc.) for both cars and their components draw attention to the eco-design phase, for which the LCA can represent an important tool.

Figure 89: Schematization of WtT, TtW, WtW and Equipment life-cycles (Nordelöf et al., 2014)

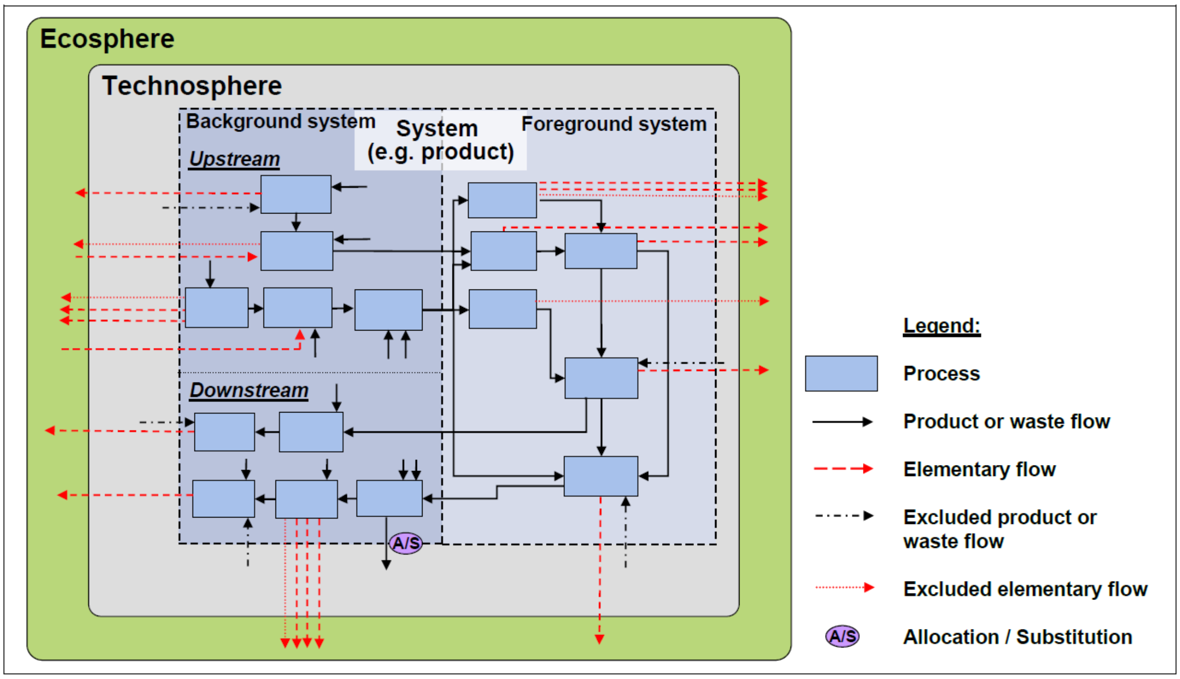



## 1.5.2 Foreground and background systems

The system under analysis could be subdivided into "background system" and "foreground system". The main difference is that the processes belonging to the background system can be assumed as representative of a process even though they don't specifically represent the analysed system. Under the management perspective, the foreground system is defined as "those processes of the system that are regarding their selection or mode of operation directly affected by decisions analysed in the study", thus directly under control of the producer. In addition, some key aspects of the EoL management could be included in the foreground system (e.g. upgradability, reusability, disassemblability, recyclability, etc.), as some design properties can directly affect the processes belonging to the foreground system. Indeed, the background processes included those processes that are not under direct control of the good producer.

The interaction between these two systems is managed by the exchange of goods and services, as they belong to the same system boundary.

Figure 90: Foreground system and background system **(EC - JRC, 2010a)**




**LCA of cars:**

The identification of the foreground and background processes for LCA applied to cars is important particularly in terms of eco-design options and improvement. This aspect can specifically refer to the in-house components and/or processes as they could be directly affected by managerial modifications. A specific focus on such components should be important in order to identify potentialities of improvement which could be achieved in short timeframes. In this case, is the components interacts with other components or with the whole vehicle, the foreground system should include this component, while the background includes all the products and services necessary for realizing the foreground. Indeed, if the entire vehicle represents the functional unit of the LCA, the complete vehicle belongs to the foreground system.

Moreover, some specific scenarios of use or EoL treatments that are investigated can become part of the foreground system with a higher level of detail.

## *1.5.3 Cut-off criteria*

**General info about LCA (from ILCD/ISO):**

All processes and flows that are attributable to the analysed system are to be included in the system boundaries. However, not all these processes and elementary flows are quantitatively relevant. For the less relevant ones, data of lower quality (estimates) can be used, limiting the effort for collecting or obtaining high quality data for those parts. Among these, the irrelevant ones can be entirely cut-off.

**LCA of cars:**

The proposed methodology implies a mass cut-off criterion comparing the weight of each component to the total weight of the car. The mass cut-off could be consistent to the International Material Data System (IMDS)[94], therefore to the Material Data Sheets (MDS) of the components. Substances which are part of each component must be reported within the MDS if their contribution by mass is higher than 0.1%, even though for some substances the threshold could be lower.

## 11.6 Preparing the basis for the impact assessment (LCIA)

### *1.6.1 LCIA methods used*

**General info about LCA (from ILCD/ISO):**

During the impact assessment analysis, the significance of potential environmental impacts using the results of the life cycle inventory analysis is assessed. In general, this process involves associating inventory data with specific environmental impacts and attempting to understand those impacts. The environmental impacts result from a complex chain of environmental mechanisms and the impact indicator can be chosen anywhere along the impact pathway (according to ISO 14044). The environmental impact of a certain product or service can be measured on a mid-point level (generally fate and exposure) or on an end-point level (fate, exposure, effect and damage). Figure 91 presents the mid-point impact (problem-oriented approach) and end-point impact (damage oriented approach) categories.

Note that the impact categories, the category indicators and the characterization models should be internationally recognised, and updates of the ILCD handbook about the recommended LCIA method are available on line (Figure ). Moreover, the category indicators should be compliant to the specific performed LCI.

[94] http://www.mdsystem.com/imdsnt/startpage/index.jsp

Figure 91: Mid-point and End-point Impact Categories (EC - JRC, 2010a)

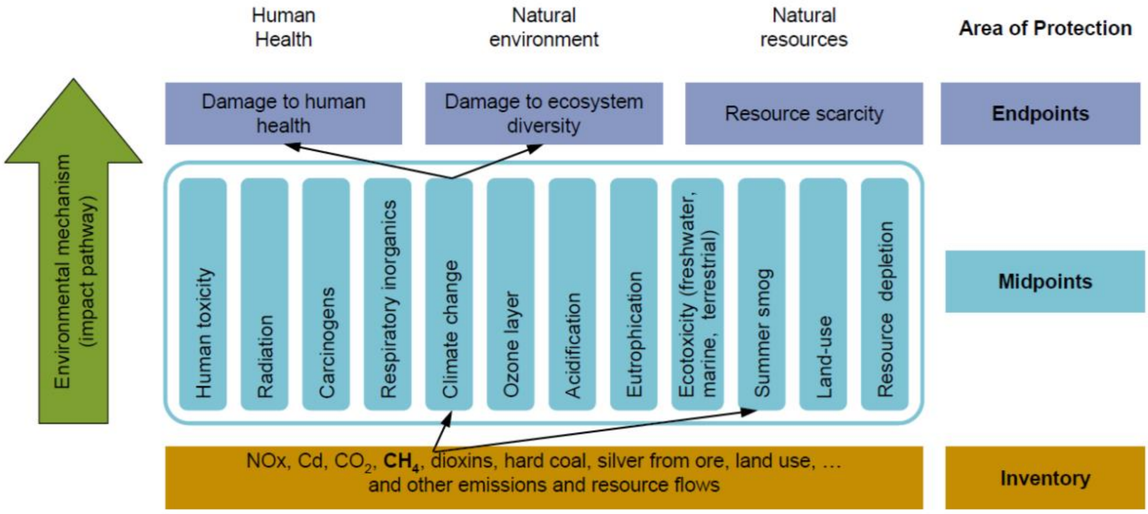
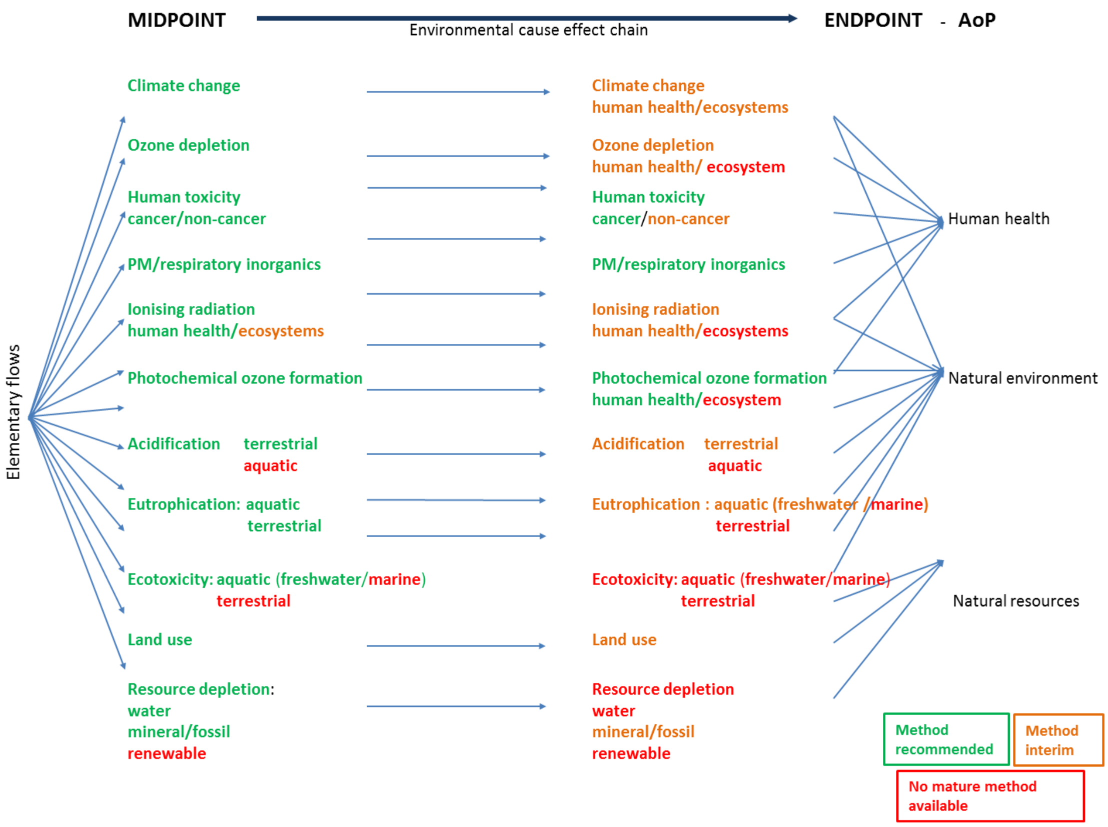



Figure 5: Overview on the recommended methods (Sala et al., 2012)



**LCA of cars:**

Overall, the recommendation is to assess the environmental impact on a mid-point level, since uncertainty is lower as compared to end-point indicators and several LCA applied to automotive report both the LCIA results.

Moreover, we recommend not to use a single indicator, but rather a broader set of mid-point indicators in order to outline the different aspects of environmental impacts and then, if necessary, a discussion of the relevance of each indicator. It is important to bear in mind that the ILCD recommended impact categories are ready-made and they are not specifically developed for the automotive sector. This means that they are not location-specific and that the LCIA outcomes could be critically interpreted based on the goal and scope of the study and the main assumptions and hypotheses adopted during the LCI (e.g. during the data collection and the establishment of the cut-off criteria). Furthermore, for critical indicators different mid-point impact assessment methods should be applied (different approach to evaluate the same impact category) in order to assess the sensitivity of the impact assessment method.

In addition, we suggest to consider also cumulated inventory result indicators, wherever the mid-point impact assessment is not yet well developed (e.g. water use impact), a mid-point impact assessment is not required (e.g. non-renewable energy demand, measured in MJ) or the emission and the resulting impacts are poorly understood (e.g. emissions from waste disposal to the ocean). Besides a quantification of the proposed cumulated inventory indicators we suggest to qualitatively discuss their environmental impacts in a second step in order to highlight potential environmental risks.

In Table 36 the LCIA methods recommended to be used within this study are presented.

Table 36: Recommended Impact Indicators to be used

| **Impact category** | **Unit** |
|---|---|
| Climate change | kg $CO_2$ equivalents |
| Acidification | Accumulated Exceedance (AE) |
| Eutrophication, terrestrial | Accumulated Exceedance (AE) |
| Ozone depletion | ODP equivalents |
| Photochemical ozone formation | kg NMVOC |
| Consumption of primary energy resources | MJ |
| Abiotic depletion (excluding primary energy depletion) | Kg Sb eq |
| Solid waste | |
| Particulate matter with diameter lower than 2.5 microns | Intake fraction for fine particles (kg PM2.5 eq/kg) |

# Life cycle Inventory data

## 12.1 Introduction

**General info about LCA (from ILCD/ISO):**

During the life cycle inventory phase, the actual data collection and modelling of the system (e.g. product) is to be done. This is to be done in line with the goal definition and meeting the requirements derived in the scope phase. The LCI results are the input to the subsequent LCIA phase.

The inventory phase involves the collection of the required data for:

- Flows to and from processes
    - Elementary flows (such as resources and emissions but also other interventions with the ecosphere such as land use),
    - Product flows (i.e. goods and services both as "product" of a process and as input/consumables) that link the analysed process with other processes,
    - Waste flows (both wastewater and solid/liquid wastes) that need to be linked with waste management processes to ensure a complete modelling of the related efforts and environmental impacts.
- Other information identified in the scope definition as relevant for the analysed system. This includes statistical data (e.g. market mix data), process and product characteristics (e.g. functions and functional units), and all other data and information, except for those directly related to impact assessment.

**LCA of cars:**

The inventory data is typically collected for the foreground system (e.g. car life cycle stages) and background data (e.g. electricity mix) are usually sourced from Automobile Industry departments and background databases such as Ecoinvent or PE databases to fill eventual gaps in collected data. However, depending on the importance of some elements and the selected impact categories, some components should be included within the foreground system.

## 12.2 Suggested approach

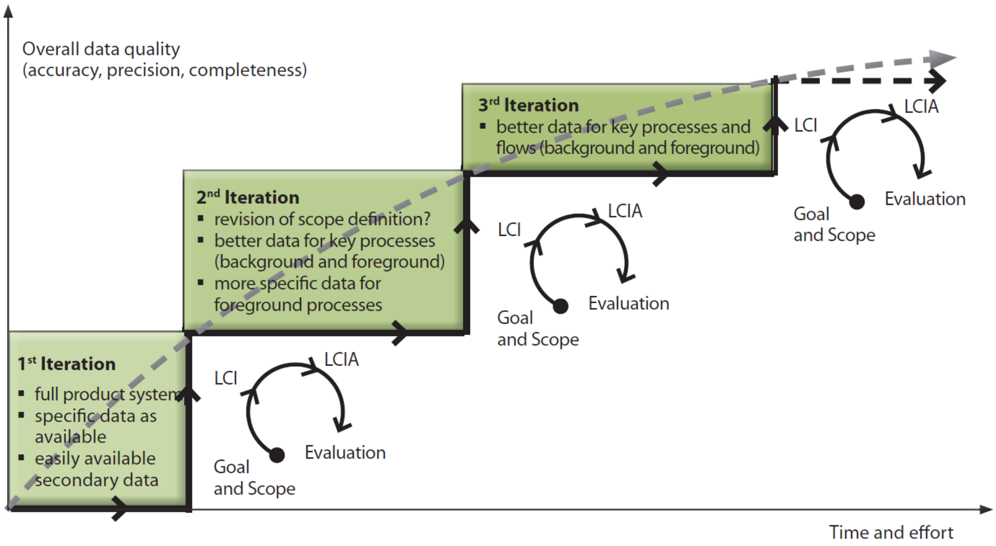
**LCA of cars:**

The establishment of high quality inventory datasets is in general very resource demanding due the several factors. Firstly, car are complex systems set up by different materials and components hailing from several suppliers around the world. Thus, the gathering of specific and detailed information is quite complicated. Moreover, the confidentiality of manufacturing information gets more difficult the obtaining of quantitative (and high qualitative) data. Even the scientific literature and the public LCA often illustrate the LCIA results but not the complete inventory, hence, making assumptions and comparison between LCIA results is not possible.

In any case, due to the complexity of the system, the use of detailed inventory data is necessary. The mass criteria should be adopted in order to fill in the inventory.

The suggestion is to adopt an iterative process permitting to make focuses and more in-depth analysis depending of the rough LCIA results and their interpretation.

Figure 92: Steps for modelling and inventory data collection




| **Definition of the LCI modelling approach:** | **Identification of foreground and background processes:** | **Collection of the foreground data:** | **Collection of the background data:** |
|---|---|---|---|
| Based on the goal and scope of the study: functional unit, system boundaries, foreground and background systems, cut-off criteria, allocation methods, data quality requirements, LCIA indicators | • Mapping of the relevant processes, Description of such components<br>• Identification of the exchanges between the foreground and the background systems | • generally through the decomposition of component or system into the relevant sub-processes in order to highlight the sources of major criticalities within the system boundaries. | • Identification of suitable data for the background processes (depending on the available information within the available databases, e.g. PE, Ecoinvent, etc.)<br>• Use of MDS from literature<br>• LCIA calculation as the base for the improvement of LCI data (iteration approach) |

# Life cycle impact assessment

Information that come from the life cycle inventory are to consider the start to evaluate environmental impacts, analysed in the phase called Life Cycle Impact Assessment (LCIA), standardized in base on ISO 14040 and 14044.

Therefore, LCIA phase includes the assignment of the LCI results to the selected impact categories and the potential environmental impacts in each category such as climate change, acidification, human health, material resource depletion, land use, etc.

The goal of the analysis of impacts is highlight importance of environmental changes caused by emissions and resources consumed because of productive processes.

LCIA uses a number of methods to convert the emission of hazardous substances and extraction of natural resources into impact category indicators at the midpoint level (such as acidification, climate change, ozone depletion and ecotoxicity), and/or impact category indicators at the endpoint level (such as damage to human health and damage to ecosystem quality), where the midpoint and endpoint level are a point positioned half way along and at the end of the environmental mechanism.

In base on ISO 14044 standard and the ILCD Handbook (EC - JRC 2012a), there are 4 steps to develop this phase:

- Selection of impact categories and classification (mandatory)

  Definition of the environmental impacts relevant to the study.

  The elementary flows from the life cycle inventory (e.g. resources consumption, emissions into air, …) are assigned to impact categories according to the substances' ability to contribute to different environmental problems.

- Characterization (mandatory)

  The impact of each emission or resource consumption is modelled quantitatively, according to the environmental mechanism. The result is expressed as an impact score in a unit common to all contributions within the impact category by applying the so called "characterization factor". For example, kg of $CO_2$ equivalents for greenhouse gases contributing to the impact category "Climate Change".

- Normalization (optional)

  The characterized impact score is associated with a common reference, such as the impact caused by one person during one year in a stated geographic context. This facilitates comparisons across impact categories and/or Areas of Protection.

- Weighting (optional)

  The different environmental impact categories and/or Areas of Protection are ranked according to their relative importance. Weighting may be necessary when trade-off situations occur in LCAs which are being used for comparing alternative products.

In following paragraphs, the most used impact categories in the automotive sector are explained.

## 13.1 Climate change

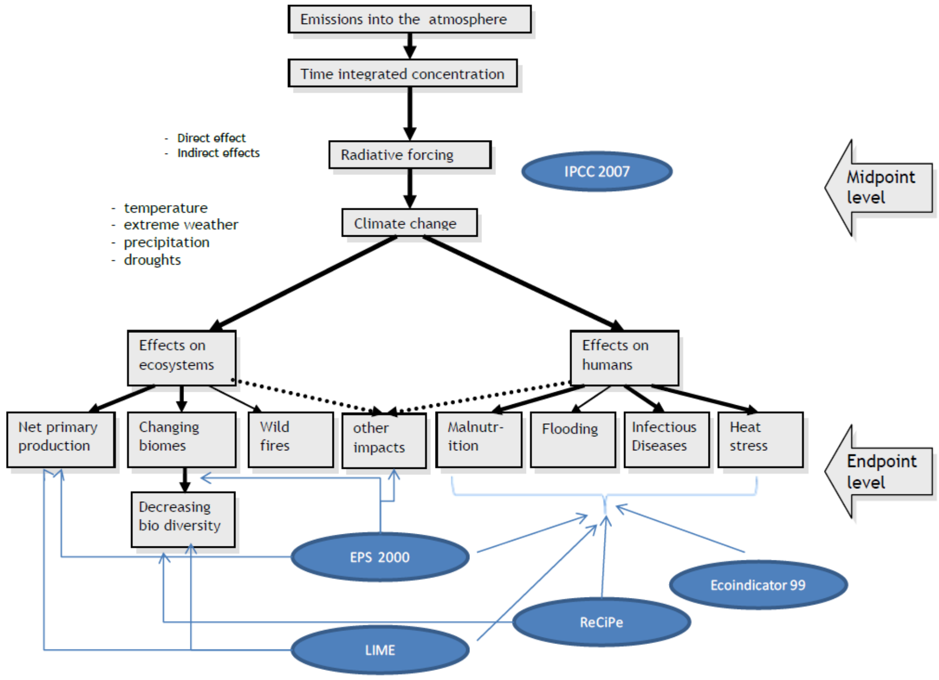
The climate change impact category is often the only impact category that is assessed in LCA studies. Moreover, several LCA studies report a partial consideration of the elementary flow contributing the overall GWP. In fact, particularly during the use-phase of the car life-cycle, the most important emissions entering in the GWP assessment are $NO_x$ and $CO_2$.

The globally recognised model for assessing the GWP has been developed by the IPCC based on the Bern model. Therefore, we suggest using the global warming potential methodology developed by IPCC with the emission factors [kg $CO_2$ equivalent] for a time horizon of 500 years (IPCC, 2006) in order to better capture all the relevant impacts of all the relevant emissions (EC - JRC, 2010b), (EC - JRC, 2011). Note that (EC - JRC, 2011) proposes as default timeframe 100 years[95], but in the same document shorter (20 years) and longer (500 years) are suggested to be used as sensitivity analysis.

In terms of endpoint indicators, the ILCD recommendations affirm that no method is considered fully matured to be recommended. In any case, although as interim method, (EC - JRC 2012b) suggest to adopt ReCiPe2008 (Goedkoop and Huijbregts, 2012)for assessing the damage on human health and ecosystem.

[95] The 100 years' timeframe is adopted as basis of the Kyoto Protocol (EC - JRC, 2011).

Figure 93: Environmental mechanism for climate change and associated LCIA methods (EC - JRC, 2011)




## 13.2 Ozone depletion

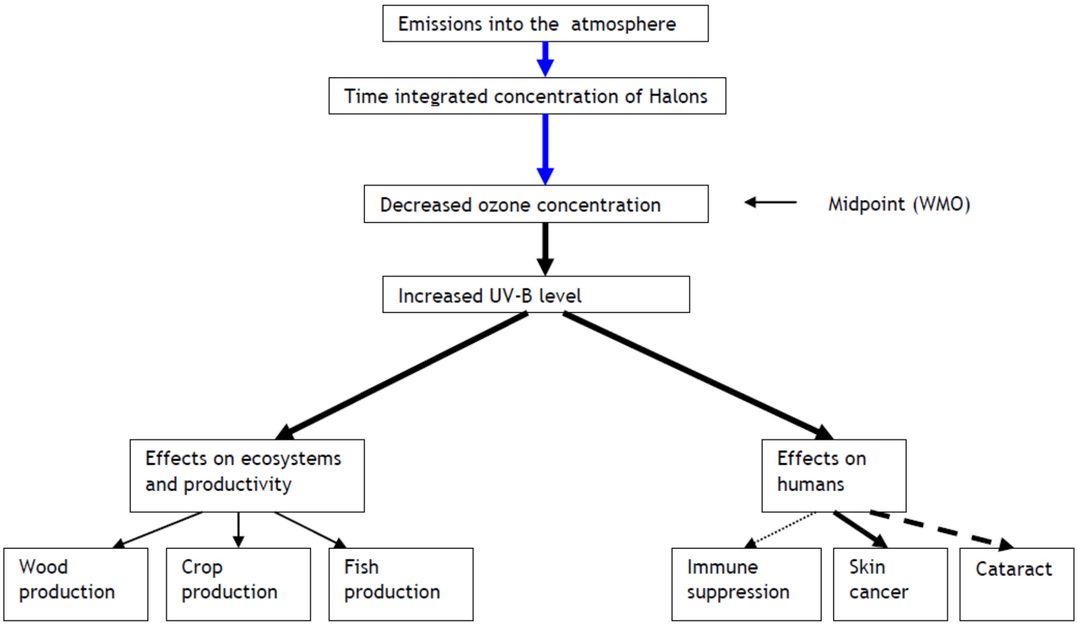
It is known that vehicles emit many reactive gases (oxidised nitrogen compounds, carbon monoxide, sulphur dioxide, volatile organic compounds) and particles that drive ozone destroying catalytic reactions.

Generally speaking, all the LCIA methodologies have an impact category assessing the Ozone Layer Depletion, that is based on the Ozone Depletion Potential (ODP) published by the WMO (World Meteorological Organization).

Thus, consistent with the ILCD recommendations, we suggest to adopt the WMO steady state method. Note that different timeframes could be used but not that the greater policy acceptance is for a 100 years perspective (EC - JRC 2011; EC - JRC 2012a).

Similarly to the Climate Change, no method are fully recommended by ILCD in terms of endpoint impact category, but the interim method that the ILCD guidelines suggest to use is that one developed by ReCiPe2008 for assessing the damage to the human health, indeed, no methods are recommended for the damage on ecosystem (EC - JRC 2012b).

Figure 94: Environmental impact pathways of ozone depletion (EC - JRC, 2011)




## 13.3 Human toxicity and eco-toxicity indicator

The assessment of human toxicity impact is strongly recommended particularly due to the existence of legislative constraints addressing the limits of toxic substances that can be emitted within the environment. Concerning LCA studies, the international recognised method for addressing the toxicity question is USEtox, which has also an almost fully compliance with the science based criteria. It is remarkable that TRACI (using the CalTOX model), has a good science-based compliance while CML2002 shows compliance just in some aspects (EC - JRC, 2011).

It is worthy that this impact category is strongly dependent of the local scale.

Therefore, the ILCD recommendation is to use the USEtox model (note that 2015 updated information are available online)[96].

**LCA of cars:**

ACEA does not recommend this type of indicator because not mature enough and heavily dependent on local background concentration levels and chemical interactions between pollutants, which can't be adequately modelled in LCA approaches. Since impact assessment methods are continuously developed by science and supported by life cycle inventory data supplier, the progress of these methods and availability of data should also be taken into account in future.

[96] http://www.usetox.org/model

## 13.4 Acidification

This impact category addresses the impacts from acidification generated by the emission of airborne acidifying chemicals. Acidification refers literally to processes that increase the acidity of water and soil systems by hydrogen ion concentration. It is caused by atmospheric deposition of acidifying substances generated largely from emissions of nitrogen oxides (NOx), sulphur dioxide ($SO_2$) and ammonia ($NH_3$), the latter contributing to acidification after it is nitrified (in the soil).

Following are presented the methods that report models for the acidification impact category at midpoint and endpoint level: TRACI acidification potentials, EDIP 2003, MEEUP, the method of Accumulated Exceedance (AE), CML 2002, ReCiPe (midpoint-endpoint method), LIME (midpoint-endpoint method) and Eco-indicator 99 (only endpoint method).

TRACI, EDIP 2003, MEEuP and LIME (at midpoint) were not recommended because they didn't reach some evaluation thresholds.

CML2002 reaches a good evaluation with the exception of being less up-to-date and showing less stakeholder importance than others. RECIPE (at midpoint) sets an interesting basis for the next generation of acidification methods based on Base saturation factor (an alternative to the critical load based methods). AE (Accumulated Exceedence) is to be preferred as default method for midpoint evaluation of acidification. This is probably the most readily adaptable method that can be used in further research to generate Global Default Characterisation Factors (CFs) or a set of consistent CFs for each continent if complemented by a set of regional/continental models which are consistent with each other (that could eventually be integrated in one global model, although not required) and expert estimate on soil sensitive area. Similar conclusions apply for CML and ReCiPe (midpoint) methods, but they both suffer from a weaker stakeholder importance. At endpoint level, no method is recommended to be use because no methods are sufficiently mature to be recommended.

## 13.5 Eutrophication, terrestrial

This impact category addresses the impacts from the macro-nutrients nitrogen and phosphorus in bio-available forms on aquatic and terrestrial ecosystems. In natural terrestrial systems, the addition of nutrients may change the species composition of the vegetation by favouring those species which benefit from higher levels of nutrients to grow faster than more nutrient efficient plants. This therefore changes the plant community from nutrient-poor (e.g. heath lands, dunes and raised bogs) to nutrient rich and more commonly, due to the widespread dispersion of nutrients, plant communities. Terrestrial eutrophication is caused by deposition of airborne emissions of nitrogen compounds like nitrogen oxides (NOx = NO and $NO_2$) from combustion processes and ammonia, $NH_3$ from agriculture.

Characterization factors for eutrophication are traditionally calculated at midpoint level, as it is the case for the majority of the LCIA methods considered in this analysis (Guinée et al., 2001), Jolliet et al. 2004, Weidema & Norris 2002). Others are damage oriented LCIA methods and relate emissions of eutrophying substances to impacts on the endpoint biodiversity (Steen 1999a; Steen 1999b; Goedkoop, M. and Spriensma, R. 2000; Payet 2006; Goedkoop et al.,2009; Itsubo 2008).

At midpoint level, methods to calculate impacts are: Accumulated Exceedance (AE), CML2002 and EDIP2003; instead, endpoint methods are: Eco-indicator 99 and EPS2000. IMPACT 2002+, LIME and ReCiPe do not include terrestrial eutrophication impacts.

As midpoint characterization method for terrestrial eutrophication it is recommended the use of the Accumulated Exceedence (Seppälä, J.et al., 2006; Posch et al., 2008) , classified as being “recommended with some improvements needed” (Level II out of III). No endpoint method is recommended for terrestrial eutrophication.

## 13.6 Photochemical ozone formation

The impact category appears under a number of different names in the various LCIA methodologies: (tropospheric) ozone formation, photochemical ozone formation or creation, photo oxidant formation, photo smog, or summer smog. There are minor differences in terms of substances included and atmospheric and meteorological conditions assumed in the modelling, but in essence they all address the impacts from ozone and other reactive oxygen compounds formed as secondary contaminants in the troposphere by the oxidation of the primary contaminants Volatile Organic Compounds (VOC) or carbon monoxide in the presence of nitrogen oxides, NOx under the influence of light. The pre-selected methods are: CML2000, EDIP2003, LIME, MEEuP, ReCiPe, TRACI (midpoint methods), EcoSense, EPS2000, LIME and ReCiPe (endpoint methods).

At midpoint and endpoint, the recommended default method is the LOTOS-EUROS model as applied in the ReCiPe method (Van Zelm, R.,2008) , is classified as Level II out of III (Recommended, some improvements needed).

The LOTOS-EUROS consists of a detailed fate and exposure model for human health impacts and is developed in a form that makes it readily adaptable for calculation of a set of consistent CFs for each continent.

## 13.7 Particulate matter with diameter lower than 2.5 microns

Ambient concentrations of particulate matter (PM) are elevated by emissions of primary and secondary particulates. The mechanism for the creation of secondary emissions involves emissions of $SO_2$ and NOx that create sulphate and nitrate aerosols. Particulate matter is measured in a variety of ways: total suspended particulates (TSP), particulate matter less than 10 microns in diameter ($PM_{10}$), particulate matter less than 2.5 microns in diameter ($PM_{2.5}$) or particulate matter less than 0.1 microns in diameter ($PM_{0.1}$).

## 13.8 Abiotic depletion

The CML method uses the Abiotic Depletion Potential (ADP), given in kg of antimony equivalents, to be multiplied with the amount of a given resource extracted. For ADP, the annual production of the resource (the extraction rate) is divided by the reserves squared, and the result divided by the same ratio for the reference resource, antimony. The value for reserves is squared to take into account the fact that a simple ratio of annual production over reserve may, in the case of higher production rates corresponding to larger reserves and vice versa, fail to reflect the impact that e.g. 1 kg of resource extraction has on overall scarcity. By including the annual production rate, CML also captures the current importance of a given resource. The CML method is recommended in the ILCD framework since it captures scarcity by including extraction as well as reserves of a given resource. Characterization factors are given for metals, fossil fuels and, in the case of reserve base and economic reserves, mineral compounds (Oers et al., 2002).In addition, the method covers most of the substances/materials identified as critical by the European Commission's Ad-hoc Working Group on defining critical raw materials. Data on reserves and production are taken from the US Geological Survey[97] . Oers et al. (2002) give characterization factors for economic reserves, reserve base, and ultimate reserves. The characterization factors given for the reserve base are recommended, as this reflects a longer time horizon and the possibility of improvement in mining technology, making feasible the exploitation of previously sub-economic deposits. The reserve base includes deposits which meet certain minimal chemical and physical requirements to potentially become economically exploitable within planning horizons (Oers et al. 2002).

[97] http://minerals.usgs.gov/minerals/pubs/mcs/

# Interpretation

## 14.1 Introduction and overview (Refers to ISO 14044:2006)

The Interpretation phase of an LCA has two main purposes that fundamentally differ:

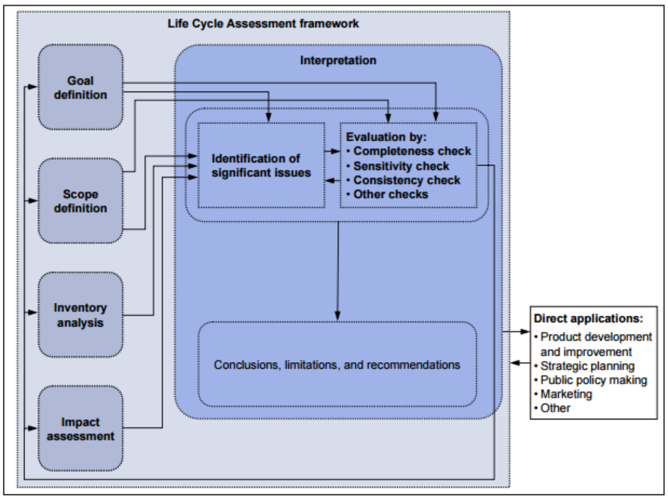
- During the iterative steps of the LCA and for all kinds of deliverables, the interpretation phase serves to steer the work towards improving the Life Cycle Inventory model to meet the needs derived from the study goal.
- If the iterative steps of the LCA have resulted in the final LCI model and results, and especially for comparative LCA studies (while partly also applicable to other types of studies), the interpretation phase serves to derive robust conclusions and - often - recommendations.

In life cycle interpretation, the results of the life cycle assessment are appraised in order to answer questions posed in the goal definition. The interpretation relates to the intended applications of the LCI/LCA study and is used to develop recommendations. The life cycle interpretation is the phase of the LCA where the results of the other phases are hence considered collectively and analysed in the light of the achieved accuracy, completeness and precision of the applied data, and the assumptions, which have been made throughout the LCI/LCA study. As said, in parallel to performing the LCI work this serves to improve the LCI model.

The interpretation proceeds through three activities as schematically illustrated in Figure 8:

- First, the significant issues (i.e. the key processes, parameters, assumptions and elementary flows) are identified.
- Then these issues are evaluated with regard to their sensitivity or influence on the overall results of the LCA. This includes and evaluation of the completeness and consistency with which the significant issues have been handled in the LCI/LCA study.
- Finally, the results of the evaluation are used in the formulation of conclusions and recommendations from the LCA study.
- In the cases where the study involves comparisons of two or more systems, additional considerations are to be included in the interpretation.

Figure 95: The element of the interpretation phase



The purpose of interpretation is to analyse and structure the results of earlier phases of the LCI/LCA study in order to identify the significant issues. There are two interrelated aspects of significant issues:

- Firstly, there are the main contributors to the LCIA results, i.e. the most relevant life cycle stages, processes and elementary flows, and the most relevant impact categories. They are important for the overall interpretation of the LCI/LCA study and for eventual recommendations. They are to be identified through a contribution analysis (also called gravity analysis), i.e. by quantifying, which contributor contributes how much to the total, resulting e.g. in stacked columns or the well-known pie charts. In the case of future scenario LCA, the contribution analysis is to be combined/build upon a scenario modelling and analysis.
- Secondly, there are the main choices that have the potential to influence the precision of the final results of the LCA. These can be methodological choices assumptions, foreground and background data used for deriving the process inventories, LCIA methods used for the impact assessment, as well as the optionally used normalisation and weighting factors. Significant choices are to be identified in a different way than the main contributors: by running the different possible choices as scenarios and comparing the scenario results.

## 14.2 Contribution analysis

Several interests and applications can require to apply the contribution analysis:

- Identify the need for further data collection or data quality improvement by quantifying the completeness of the inventory.
- Focus further data collection efforts on the most contributing processes and individual elementary flow interventions.
- Focus efforts in ecodesign and product improvement / development on the most contributing processes and individual elementary flow interventions.
- Communicate the share of internal vs. external contribution to the overall environmental impact in context of customer or stakeholder communication.
- Contribute to internal quality control during the LCA work by investigating the qualitative and quantitative plausibility of the detailed outcome of the contribution analysis; this is part of the interim and final evaluation of the LCI/LCA study results.

Depending on the drivers, inventory data-related significant issues are to be identified among whole life cycle stages, producer internal / external processes, groups of activities (e.g. transportation, energy production, services), key processes, and/or key elementary flows / interventions. If key processes of the system are parameterised, these parameters can equally be significant issues. The analysis is typically done on multiple levels, e.g. for LCIA results: first in relation to the individual elementary flows, secondly in relation to the individual impact categories on midpoint and/or category endpoints on endpoint level, and thirdly in relation to the overall (normalised and weighted) environmental impact. The third step is in general also called dominance analysis. In practice the contribution analysis is supported by professional LCA tools, or can be done by analysis of the inventory and LCIA result tables in spreadsheet software.

## References


Chlopek, Z., Lasocki, J., 2011. Comprehensive evaluation of the environmental hazard caused by the operation of automotive vehicles. Arch. Motoryz. Arch. Automot. Eng.

Directive 1999/94/EC, 1999. This document is meant purely as a documentation tool and the institutions do not assume any liability for its contents. Off. J. Eur. Communities L 269, 1–15. doi:2004R0726 - v.7 of 05.06.2013

Directive 2009/33/CE, 2009. 15.5.2009. Off. J. Eur. Communities.

EC, 2003. Communication COM(2003:302) - Integrated Product Policy - Building on Environmental Life-Cycle Thinking. COM(2003) 302 final.

EC - JRC, 2012. Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. JRC-EC (Joint Research Centre - European Commission). doi:10.2788/60825

EC - JRC, 2011. ILCD Handbook: Recommendations for Life Cycle Impact Assessment in the European context. doi:10.278/33030

EC - JRC, 2010a. International Reference Life Cycle Data System (ILCD) Handbook - Framework and Requirements for Life Cycle Impact Assessment Models and Indicators. Publications Office of the European Union, European Commission - Joint Research Centre - Institute for Environment and Sustainability. doi:10.2788/38479

EC - JRC, 2010b. ILCD Handbook: Framework and requirements for LCIA models and indicators. doi:10.2788/38719

EU, 2014. REGULATION (EU) No 333/2014 OF THE EUROPEAN PARLIAMENT AND OF THE COUNCIL - the modalities for reaching the 2020 target to reduce CO2 emissions from new passenger cars.

Goedkoop, M., Heijungs, R., Huijbregts, M.A.J., De Schryver, A., Struijs, J., V.Z., R., 2009. ReCiPe 2008 A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. Report I: Characterisation factors, first edition.

Goedkoop, M. and Spriensma, R. (2000). Ministry of VROM, T., Hague, T.N. (updated version: www. pre. nl/eco-indicator9. ), 2000. No TThe Eco-indicator 99: A damage oriented method for Life Cycle Impact Assessment Methodology.itle.

Goedkoop, M., Huijbregts, M., 2012. ReCiPe 2008.

Guinée, J.B., Gorrée, M., Heijungs, R., Huppes, G., Kleijn, R., de Koning, A., Van Oers, L., Anneke, W.S., Suh, S., De Haes, U.H.A., De Bruijn, H., Van Duin, R., Huijbregts, M.A.J., 2001. Handbook on Life Cycle Assessment. Operational Guide to the ISO Standards, 2001. Centre of Environmental Science. Leiden University, School of Systems Engineering, Policy Analysis and Management – Delft University of Technology, Fuels and Raw Materials Bureau, Interfaculty Department of Environmental Science, University of Amsterdam, Department of Environmental Studi.

IPCC, 2006. 2006 IPCC Guidelines for National Greenhouse Gas Inventories 2, 12. doi:http://www.ipcc-nggip.iges.or.jp/public/2006gl/pdf/2_Volume2/V2_3_Ch3_Mobile_Combustion.pdf

Itsubo, N. et al. (2008a), 2008. LIME documentation. Eutrophication. Chapter 2.8 of draft English translation of LIME documentation (figures and tables in Japanese).

Jolliet, O., Müller-Wenk, R., Bare, J., Brent, A., Goedkoop, M., Heijungs, R., Itsubo, N., Peña, C., Pennington, D., Potting, J., Rebitzer, G., Stewart, M., Haes, H.U., Weidema, B., 2004. The LCIA midpoint-damage framework of the UNEP/SETAC life cycle initiative. Int. J. Life Cycle Assess. 9, 394–404. doi:10.1007/BF02979083

Messagie, M., Boureima, F.S., Coosemans, T., Macharis, C., Mierlo, J. Van, 2014. A range-based vehicle life cycle assessment incorporating variability in the environmental assessment of different vehicle technologies and fuels. Energies 7, 1467–1482. doi:10.3390/en7031467

Molenbroek, E., Edith Molenbroek, Smith, M., Groenenberg, H., Waide, P., Attali, S., Fischer, C., Krivošik, J., Fonseca, P., Santos, B., Fong, J., 2014. Evaluation of Energy Labelling Directive and specific aspects of the Ecodesign Directive.

Nemry, F., Leduc, G., Mongelli, I., Uihlein, A., 2008. Environmental Improvement of Passenger Cars

(IMPRO-car). European Commission, Joint Research Centre (JRC), Institute for Prospective Technological Studies (ipts). doi:10.2791/63451
Nordelöf, A., Messagie, M., Tillman, A.M., Ljunggren Söderman, M., Van Mierlo, J., 2014. Environmental impacts of hybrid, plug-in hybrid, and battery electric vehicles-what can we learn from life cycle assessment? Int. J. Life Cycle Assess. 19, 1866–1890. doi:10.1007/s11367-014-0788-0
Oers, L. van, A. de Koning, J.B. Guinée & G. Huppes, 2002. Abiotic resource depletion in LCA - Improving characterisation factors for abiotic resource depletion as recommended in the new Dutch LCA Handbook. DWW report, Delft.
Payet, J. (2006)., 2006. Deliverable D.4.1.4 Report describing a method for the quantification of impacts on aquatic freshwater ecosystems resulting from different stressors (e.g., toxic substances, eutrophication, etc).
Posch, M., Seppälä, J., Hettelingh, J.P., Johansson, M., Margni M., Jolliet, O., 2008. The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA.
Sala, S., Pant, R., Hauschild, M., Pennington, D., 2012. Research needs and challenges from science to decision support. Lesson learnt from the development of the international reference life cycle data system (ILCD) recommendations for life cycle impact assessment. Sustainability 4, 1412–1425. doi:10.3390/su4071412
Seppälä, J., Posch, M., Johansson, M., Hettelingh, J.P., 2006. Country-dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator.
Steen, B., 1999a. A Systematic approach to environmental priority strategies in product development (EPS). Version 2000-Models and data of the default method; CPM report 1999:5.
Steen, B., 1999b. Environmental Assessment of Products and Material Systems A systematic approach to environmental priority strategies in product development ( EPS ). Version 2000 – General system characteristics 66.
The, Q., Council, Q., Affairs, E., n.d. EU Sustainable Consumption and Production Action Plan The Quaker Council for European Affairs.
Van Zelm, R., Huijbregts, M.A.J., Den Hollander, H.A., Van Jaarsveld, H.A., S., F.J., Struijs, J., Van Wijnen, H.J., Van de Meent, D., 2008. European characterization factors for human health damage of PM10 and ozone in life cycle impact assessment. Atmos. Environ. 42, 441-453.
Weidema, B.P., Norris, G. a, 2002. Avoiding co-product allocation in the metals sector. ICMM Int. Work. Life Cycle Assess. Met.

# Appendix B

Table 37: Life-cycle Impact Assessment of the two FCA models (*B segment gasoline 1* and *B segment gasoline 2*) (ILCD/PEF recommended impact categories)

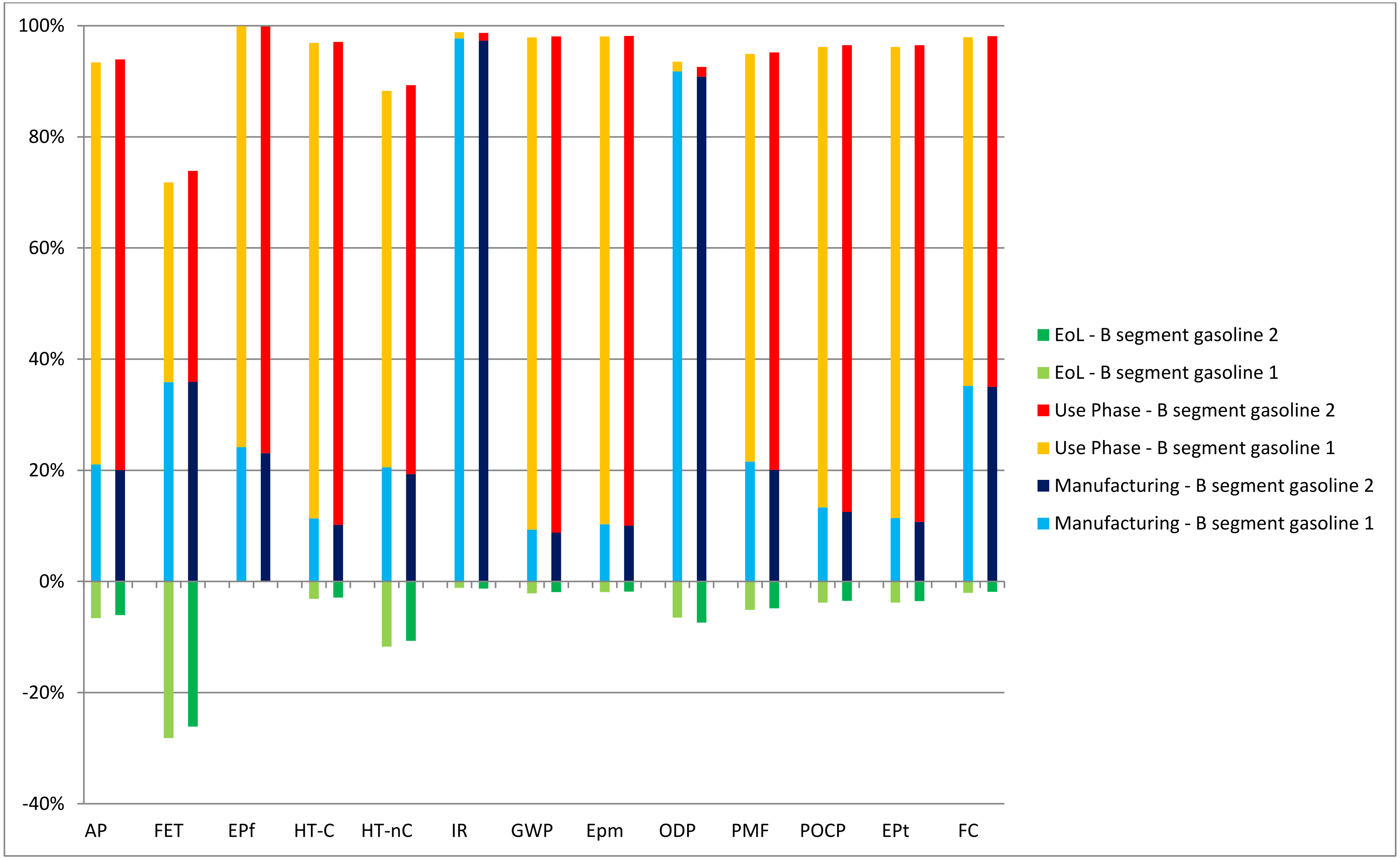
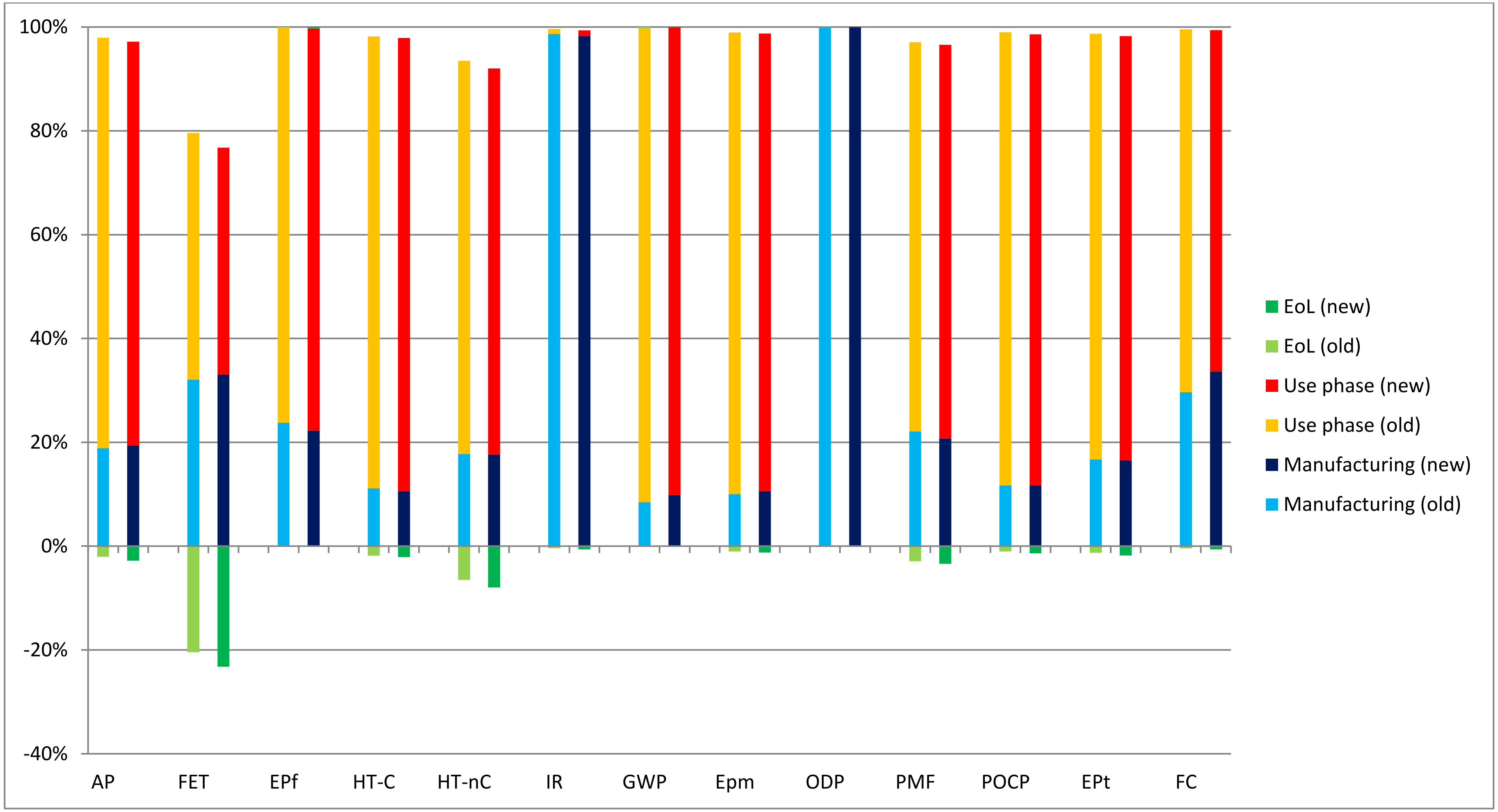


Table 38: Life-cycle Impact Assessment of the two models of *A segment gasoline* (ILCD/PEF recommended impact categories)



# Appendix C

Figure 96: Contribution of the EoL treatments compared to the EoL environmental impact for the *A segment gasoline* vehicles (100% = EoL environmental impact)

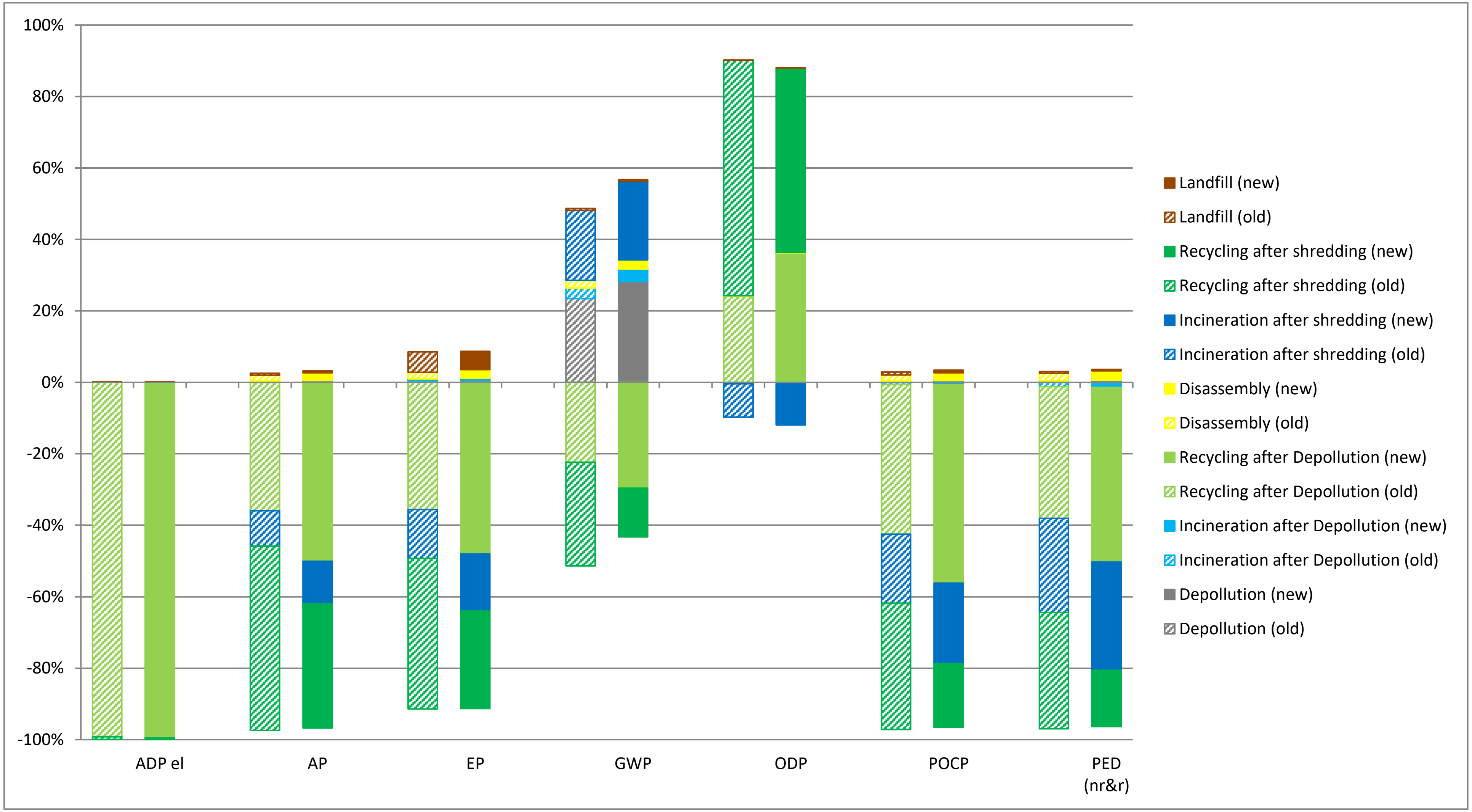


Figure 97: Contribution of the EoL treatments compared to the EoL environmental impact for the *B segment gasoline 1* and *B segment gasoline 2* vehicles (100% = EoL environmental impact)

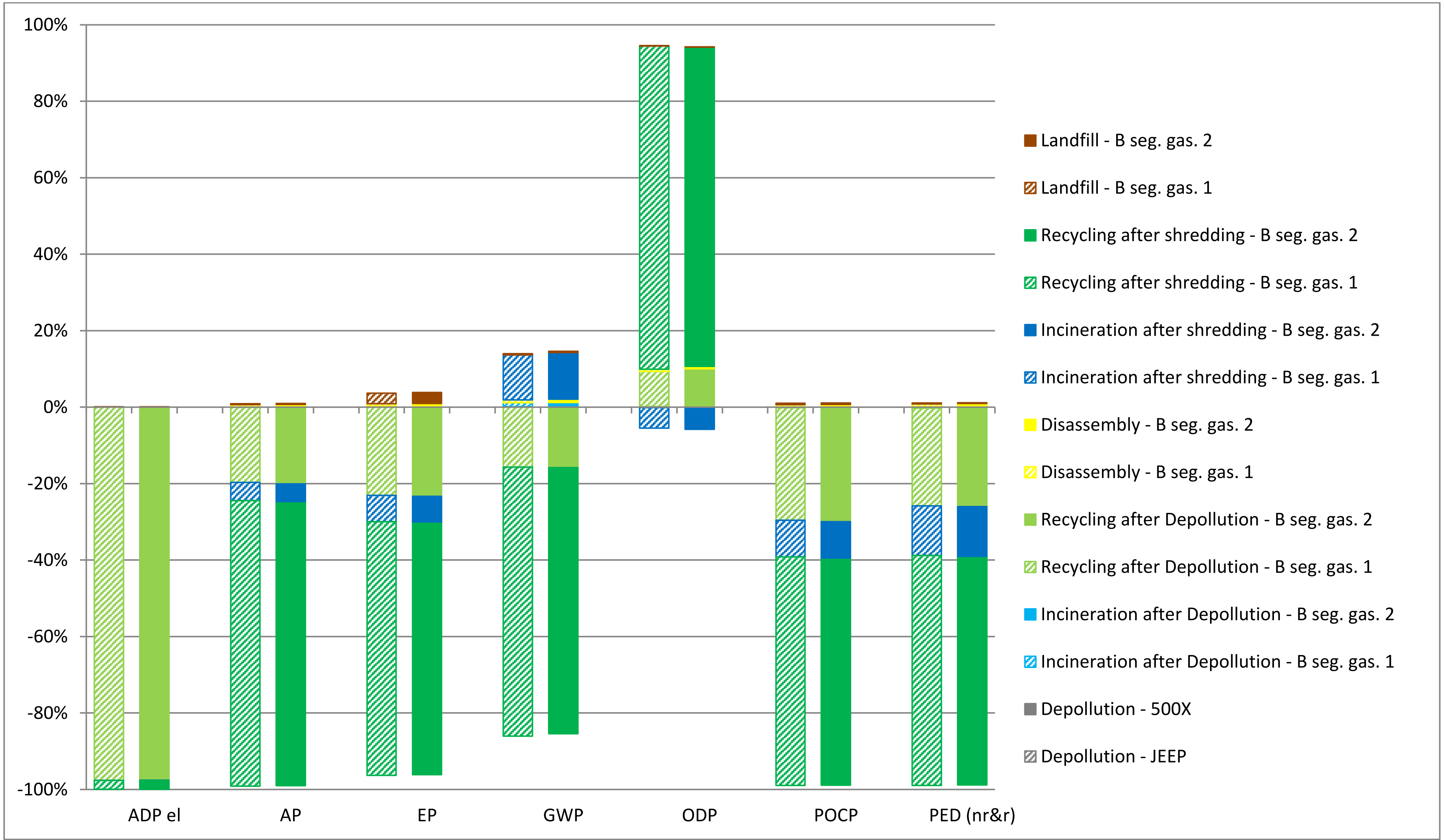


Figure 98: Contribution of the EoL treatments compared to the EoL environmental impact for the A segment gasoline 1 vehicles (100% = EoL environmental impact)

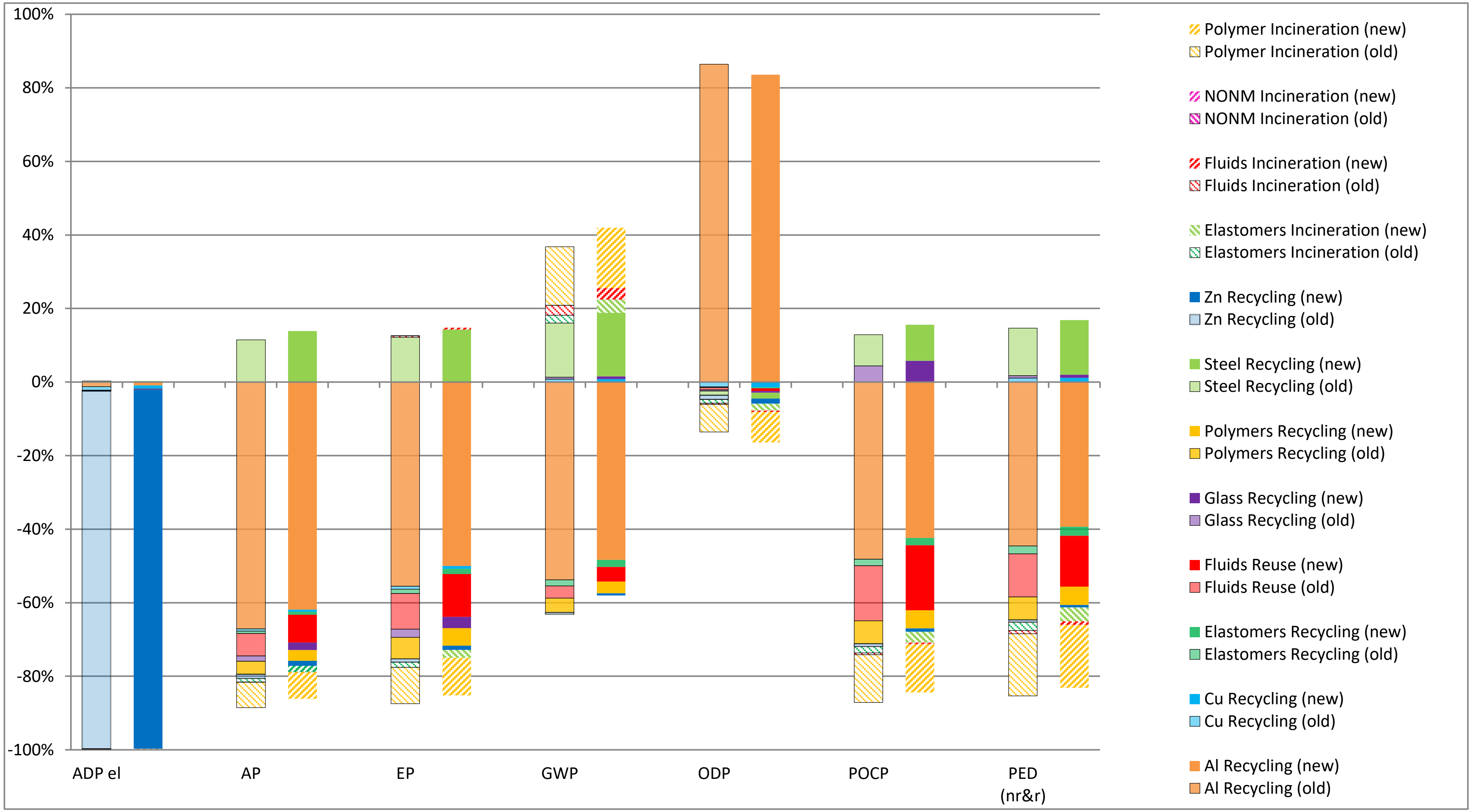


Figure 99: Contribution of the EoL treatments compared to the EoL environmental impact B segment gasoline 1 and B segment gasoline 2 vehicles (100% = EoL environmental impact)